DOCTOR OF PHILOSOPHY

Mapping and Assessment of changes in ecosystem service delivery - a historical perspective on the Tweed catchment, Scotland - UK

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Mapping and Assessment of changes in ecosystem service delivery – a historical perspective on the Tweed catchment, Scotland - UK

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Declaration

I, **Sikhululekile Ncube**, declare that I am the author of this thesis; that, unless otherwise stated, all references cited have been consulted by me; that the work of which the thesis is a record has been done by me and that it has not been previously accepted for a higher degree.

Signature…………………. Date: ……………

Abstract

For centuries, river catchments and their constituent habitats have been altered and modified through various human activities to maximise provision of tangible benefits like food and water, while impacting on their capacity to provide other less obvious but equally important benefits for human survival. However, in the last few decades, perceptions on the role of catchments as mere providers of tangible benefits have been changing, as recognition has been given to other human beneficial services like regulation of floods. This recognition has drawn increased interest in both science and policy, towards understanding human-nature relations and how approaches like the ecosystem services concept can inform sustainable management of catchments.

Although, the multiple and differently weighted relationships existing between habitats and ecosystem services have been acknowledged, the relationship between spatio-temporal change in habitats and spatio-temporal change in ecosystem services delivery, has not received as much attention in the research literature. In this thesis, it is argued that this is an important omission as spatio-temporal habitat change could have broader consequences for ecosystem services provided by a catchment. On this basis, this study maps and assesses the influence of habitat changes across space and time on ecosystem services delivery at a local catchment scale.

Approaches to assessing ecosystem service delivery across landscapes and catchments draw on habitat mapping data for those landscapes or catchments. Such data are in turn used as proxies for estimating different ecosystem services delivered by the landscape or catchment based on their integration with other spatial or non-spatial data. To date this approach has been applied to assess contemporary delivery of different ecosystem services. The basis of the approach taken in this study involved comparing a pre-existing contemporary ecosystem service assessment of two chosen sub catchments of the Tweed catchment in Scotland, with a similar assessment based on a set of older “historic” habitat maps for the mid-20th century period. Derivation of the digital map base for the latter was a major focus of the present study.

Aerial photography taken during the Royal Air Force surveys in the 1940s archived in the Royal Commission on Ancient and Historical Monuments of Scotland were obtained and first scanned digitally, arranged into a mosaic of adjacent images and ortho-rectified to remove camera distortion. These photo mosaics were then visually interpreted and, aided with ancillary data, the current (2009) habitat maps were edited and backdated to derive the historic habitat maps for the study catchments. The Spatial Evidence for Natural Capital Evaluation (SENCE) ecosystem services mapping approach was then used to translate generated habitat maps into ecosystem service supply maps.

Findings show that the study catchments changed from multifunctional to intensively managed landscapes by 2009, with a higher capacity for supplying provisioning ecosystem services, while their capacity to supply regulating and supporting ecosystem services was reduced. Findings also show that a change in one habitat type results in changes in multiple ecosystem services, while changes in the spatial configuration of habitats reduces areas with high supply capacity for regulating and supporting ecosystem services. This study concludes that ecosystem service delivery is not only affected by changes in gross area of constituent habitats but also by spatial changes in the configuration and distribution of these habitats. In this regard, it is argued that recognising and understanding changes in ecosystem services adds an important strand in catchment management. It is therefore suggested that planning for future ecosystem services in catchment management needs to be informed by historic baselines.

Acronyms

AAG	Area Advisory Group
CAP	Common Agricultural Policy
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
CORINE	Coordination of Information on the Environment
DEM	Digital Elevation Model
DPSIR	Driver Pressure State Impact Response
DTM	Digital Terrain Model
EA	Ecosystem Approach
ES	Ecosystem Services
ESA	Ecosystem Services Approach
EU	European Union
FESP	Framework for Ecosystem Service Provision
GCP	Ground Control Point
GIS	Geographical Information Systems
HELP	Hydrology for the Environment, Life and Policy
IACS	Integrated Administration and Control System
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs
IWRM	Integrated Water Resources Management
JNCC	Joint Nature Conservation Committee
LC/LU	Land Cover/Land Use
LCS88	Land Cover Scotland 1988
LUS	Land Use Strategy
MAES	Mapping and Assessment of Ecosystem Services
MEA	Millennium Ecosystem Assessment
NFM	Natural Flood Management
NGO	Non-Governmental Organisation

OS	Ordnance Survey
RAF	Royal Air Force
RBMP	River Basin Management Plans
RCAHMS	Royal Commission on the Ancient and Historic Monuments of Scotland
SAGA	System for Automated Geoscientific Analyses
SBC	Scottish Borders Council
SENCE	Spatial Evidence for Natural Capital Evaluation
SEPA	Scottish Environment Protection Agency
SFM	Structure from Motion
SNH	Scottish Natural Heritage
SSSI	Site of Special Scientific Interest
TEEB	The Economics of Ecosystems and Biodiversity
UK-NEA	United Kingdom National Ecosystem Assessment
UN	United Nations
UNESCO	United Nations Educational, Scientific and Cultural Organisation
WEWS	Water Environment and Water Services
WFD	Water Framework Directive
WTA	Willingness to Accept
WTP	Willingness to Pay

1 Introduction

1.1 Chapter introduction

While the multiple and differently weighted relationships between habitats and ecosystem services have been acknowledged (Maes et al., 2013), the relationship between spatio-temporal changes in habitats and spatio-temporal changes in ecosystem service delivery has, so far received little attention in literature (Haines-Young et al., 2012). In this thesis, it is argued that this is an important strand in understanding ecosystem services as this might imply that spatio-temporal habitat changes could have broader consequences for ecosystem services provided by a catchment. In this regard, this chapter presents the background to this study. It first provides a general overview on river catchments as important sources of multiple ecosystem services needed for human survival, but which due to years of human induced modifications and changes, are among the most threatened and degraded ecosystems. It also briefly discusses increasing interest in understanding human-nature relations through emerging notions like the ecosystem services concept and its use in the management of river catchments and the wider natural environment. The second section provides the rationale for this study and knowledge gaps informing the aim of this study. The last section presents an outline of the structure of this thesis, providing a brief highlight of what each of the chapters encompass.

1.2 Background and Context setting

Worldwide, river catchments and their constituent habitats provide a number of multiple benefits needed for human survival (Newson, 1996, Tané, 1996), referred to as ecosystem services. The Millennium Ecosystem Assessment (2005) defined ecosystem services as "*the benefits people obtain from ecosystems*." These are categorised into four broad categories of provisioning, regulating, cultural and supporting ecosystem services. One of the most vital benefits provided by catchments is water for among other uses, agricultural production, domestic supply, hydropower and industrial uses. River catchments also serve as centres of socioeconomic development in both developed and developing countries.

In the developed world, river catchments have emerged as centres for intensive agriculture and infrastructural development. For example, in Scotland, salmon fishing in the Tweed catchment provides about 500 jobs and significantly contributes about £18 million/year to the local economy (The Tweed Foundation, 2014). Equally, in the developing world, many communities entirely rely on river catchments as sources of livelihood and for household sustenance (Maltby and Acreman, 2011,Verma and Negandhi, 2011, Russi et al., 2013). For example, major river basins in Africa such as the Zambezi play a fundamental role in supporting livelihoods through providing fertile soils for agriculture, fish, timber, medicines and other raw materials like reeds (Ncube et al., 2013).

Catchments also provide an array of less tangible benefits which are equally important for human survival. These include cultural services such as recreation, tourism, aesthetic value, religious beliefs, education, historical, archaeological sites and cultural heritage (Everard, 2013, Ferrier and Jenkins, 2010). For example, the peat and clay deposits in the Somerset Levels and Moors wetlands in England contain plant and animal remains such as pollen, seeds, snails and beetles and these provide records on changing climate, sea level rise and landscape overtime (Acreman et al., 2011). In Bhopal (India) for example, the Bhoj wetlands are used for the immersion of idols and other religious icons during the Hindu and Muslim religious festivals (Verma and Negandhi, 2011) and the Ganges River (India) is considered a sacred place as it is associated with religious and spiritual beliefs.

In their natural state, river catchments also play a crucial role in a number of regulation services such as flood control, carbon storage, water purification and nutrient removal (Acreman et al., 2011, UK-National Ecosystem Assesment, 2011). They also support diverse plant and animal species including birds, invertebrates, wildlife and varied microbial communities (Blackwell and Pilgrim, 2011).

Despite their importance, river catchments are among the most threatened and degraded ecosystems (Febria et al., 2015, Gilvear et al., 2013), subjected to a number of pressures. Over 60% of the world's ecosystem services including river catchments were reported to be degraded or exploited in an unsustainable way (Millennium Ecosystem Assessment, 2005). This was largely attributed to modification and alteration of rivers flows, quality and structure including their catchment areas (Gilvear et al., 2013, Robinson, 1990), through a range of human activities aimed at tangible benefits like food provision and water supply.

For centuries, river catchments and their constituent habitats have been altered and modified in different parts of the world (Newson, 1997). Agriculture activities, engineered flood control measures and dam constructions are among the dominant human activities that have led to marked alteration and degradation of river catchments. Estimates show that, since the beginning of the 20^{th} century the world has lost 50% of its wetlands, mainly attributed to drainage of these for agricultural purposes while about half of the large river systems worldwide are altered by dams (Nilsson et al., 2005). The UK for example, lost over 90% of the original wetland extent due to drainage of these areas for agricultural production (UK-National Ecosystem Assesment, 2011). Such agricultural activities in turn resulted in pressures such as diffuse pollution and habitat degradation.

Straightening of rivers and dam constructions detach main river systems from their adjacent land including their floodplains and this impacts on the flow regime, flooding and inundation processes responsible for exchange of nutrients, sediments and floodplain replenishment (Junk et al., 1989, Thoms et al., 2005, Ward, 1998). Australia for example, lost about 90% of its floodplains through regulation of rivers (Schofield et al., 2003, Thoms, 2003) while 13% of floodplains in the UK were degraded or completely disconnected from river channels (Hume, 2008). In so doing, the ability of river catchments to regulate processes like flood control is impacted upon including the multi-benefits they provide (Scottish Environment Protection Agency, 2015a).

Such degrading impacts of human activities in river catchments have partially been attributed to lack of appreciation or interest in the multiple benefits these provide other than the tangible provisioning ecosystem services of water supply and food provision (Rodríguez et al., 2006). On the other hand, others like Verma and Negandhi (2011) are of the view that it is actually due to these multi benefits that river catchments are subjected to many decision makers with differing interests and priorities. Differing preferences, priorities and values, some of which are incompatible present different layers of complexity in catchment management as these operate at different scales (Ferrier and Jenkins, 2010). Addressing these discrepancies requires different disciplines and approaches to integrate social values and priorities in managing catchments. Deliberations among competing users also often become intermingled with politics as decision makers alter and modify catchments to maximise the provision of prioritised benefits (Zeitoun et al., 2013). Such discrepancies and varying preferences present what

Balint et al. (2011) refer to as “wicked” environmental problems, as there is no single ideal and best solution to balancing the multi benefits in catchment management.

In the last few decades, perceptions on the role of river catchments as mere providers of tangible benefits have been changing as recognition has also been given to other equally human beneficial services like flood regulation. This is attributed to the illustration by the Millennium Ecosystem Assessment (2005) on the multiple benefits provided by the natural environment and how these contribute to human well-being, which further raised awareness and the impetus for understanding human-nature relations (Mulder et al., 2015, Daily, 1997).

In the recent past, the impacts of human activities have emerged as threats to both humanity and biodiversity worldwide (Millennium Ecosystem Assessment, 2005). In the UK, for example, with its history of catchment alterations and modifications (river channelisation, wetland drainage, intensive agriculture and land use changes), pressures such as flooding, diffuse pollution, abstraction etc. have been on the increase (European Union, 2000, Scottish Environment Protection Agency, 2015b).

In recognition of these challenges, governments across the world have increased attention towards understanding human-nature relations and management approaches that work “with”, rather than “against” nature (Guerry et al., 2015, Schaefer et al., 2015). Consequently, ecosystem based and integrated approaches like the ecosystem services concept have gained increasing interest in both policy and science (UK-National Ecosystem Assesment, 2011). The ecosystem service concept focusses on understanding the multiple benefits provided by the natural environment to humans and the need to manage the natural environment so that the provision of these multiple benefits is sustained over long term (Cook and Spray, 2012). It is viewed as a bridging concept between science, policy and practice, providing a conceptual framework for the management of the natural environment (Plant and Ryan, 2013, Schröter et al., 2014). However, there are calls for more understanding of this concept, including how it could inform management of catchments or add value in implementation of other policy intentions such as the EU Water Framework Directive in river basin planning and management (Wallis et al., 2011, Blackstock et al., 2015).

1.3 Rationale, scope and aims of this study

1.3.1 Rationale and scope of this study

This study focusses on catchments, as important ecosystems with an array of habitats which, although they provide a range of benefits, have been subjected to historic alterations and modifications. Two sub catchments (Ale and Eddleston) of the Tweed catchment in Scotland (UK) have been selected to provide empirical evidence of assessment of spatio-temporal changes in ecosystem services. The selected study catchments are typical of the Scottish (Harrison, 2012) and arguably the wider British countryside subjected to historic river system alterations and land use changes related to intensive agriculture and infrastructural development. Of consideration was also the catchment scale at which ecosystem services are mapped, as local scales are increasingly considered as important for decision making and environmental policy implementation (Spray and Blackstock, 2013).

This study specifically focusses on mapping and assessing changes in past ecosystem services as a basis for understanding the current state of ecosystem service delivery in the catchments, how these have changed over time and how such an understanding can inform ecosystem service delivery and catchment management in future. Haines-Young et al. (2012) note the need to describe changes in ecosystem services resulting from habitat/land cover modifications and changes, as most studies have focussed on contemporary delivery of different ecosystem services. For example, Metzger et al. (2006) used the present/current as a baseline to inform future ecosystem services delivery and management, with limited knowledge about past ecosystem services. Other studies e.g. Raudsepp-Hearne et al. (2010) and Maes et al. (2012b), have focussed on assessment of contemporary ecosystem service interactions across space, with limited focus on assessing changes to these over time. Yet, as argued by Johansson et al. (2008), explaining present ecosystems without historic understanding provides a limited perspective on causes of ecosystem changes. Understanding past ecosystem services can help explain ongoing ecosystem changes and help put the current situation into context (Swetnam et al., 1999).

In the recent past, ideas on assessment of ecosystem services through mapping have been developing as indicated by an increase in studies focusing on mapping ecosystem services (Crossman et al., 2013). As a consequence, mapping ecosystem services is at the core of current contemporary approaches to assessing ecosystem services delivery across landscapes or catchments. By mapping ecosystem services both tangible and less obvious ecosystem services are explicitly spatially presented and in so doing, this makes "invisible" ecosystem services "visible". This further allows for visualisation of ecosystem service supply areas, as well as understanding spatial variations in ecosystem service supply within catchments (Brauman et al., 2014). In addition, ecosystem service interactions e.g. trade-offs and synergies across space and time can also be analysed.

Mapping ecosystem services has been identified as a key element of the ecosystem services concept which could contribute towards moving this concept into practice (Daily and Matson, 2008) by providing the science evidence base needed to underpin policy implementation (Brauman et al., 2014). This is particularly in view of increasing concerns over the implementation of the ecosystem services concept (Nahlik et al., 2012, Norgaard, 2010) and calls for more understanding on how this concept can move into practice (Bouma and Beukering, 2015). Although significant progress has been made on understanding the ecosystem services concept (Potschin and Haines-Young, 2011), mapping ecosystem services has been the subject of relatively little research (Andrew et al., 2015), as most studies have focussed on other aspects such as attempts to define ecosystem services and economic valuation of ecosystem services.

1.3.2 Aims of this study

The aim of this study is to assess spatio-temporal changes in ecosystem service delivery in two catchments by utilising and extending the ecosystem service mapping technique based on habitat maps, and to consider how this approach may help inform catchment management.

As reflected in the aforementioned aim, this study first sought to understand habitat changes as informing changes in ecosystem service delivery. This is because currently established practices of mapping ecosystem services rely on proxy based approaches due to lack of independent ecosystem services data (Burkhard et al., 2012). Among these, the use of habitat/land cover data is the most commonly used approach (Seppelt et al., 2011). As will be discussed later, different habitat/land cover types are equated to different ecosystems which produce different combinations of ecosystem services. The classification of habitat types is argued to significantly overlap with that of ecosystems (Maes et al., 2013). Habitats are primary landscape units providing ecosystem services due to their dominance in the landscape (Bolliger and Kienast, 2010, Syrbe and Walz, 2012).

To address the aim of this study, research questions that sought to understand spatio-temporal changes in habitats and how these in turn influenced changes in ecosystem services delivery were posed as:

Research questions:

1. What is the historic state and patterns of habitat change in the Ale and Eddleston catchments?
2. What are the historic ecosystem services in these catchments and what changes have occurred to these?
3. How have the identified changes in habitats influenced ecosystem service delivery in both catchments?
4. What are the similarities and differences between habitat change and ecosystem service delivery across both catchments?
5. Which key factors influenced habitat and ecosystem service changes in the study catchments?

The results and discussion chapters provide responses to these research questions.

1.4 Thesis structure

This thesis is presented in six chapters. Below is a brief summary of each of these chapters.

Chapter 1: Introduction – This chapter provides the background to this study, highlighting the importance of river catchments as well as the threats and pressures faced by these. It also provides an overview of the emerging focus on the possible use of ‘notions’ such as ecosystem services to understand human-nature relations and how these can potentially add value to environmental policy implementation through informing the management of ecosystems like river catchments. Also included in this chapter was the rationale and scope of this study.

Chapter 2: Literature review – The aim of this chapter is to collate and analyze literature covering a general understanding of the ecosystem service concept, mapping and assessment of changes in these. It also highlights the increased focus on catchment management using such ecosystem-based approaches. It is divided into three main sections. The first section discusses the importance of catchments as suitable units for environmental management and policy implementation, including discussions on criticisms levelled against earlier notions of integrated catchment management and the emerging focus on the possibilities of the use of the ecosystem service concept in catchment management. Section two discusses the definition of ecosystem services, its origins and classification; highlighting the areas of debate surrounding this concept, as well as introducing conceptual frameworks to understand this concept. Section three reviews literature on mapping ecosystem services, including methods currently in use, the tools that have been developed, types of ecosystem services frequently mapped, and the scales at which they are mapped. Importantly, it also discusses current approaches used to assess changes in ecosystem service delivery over time.

Chapter 3: Methodology – This chapter presents the data collection and analysis methods used in this study. The first section presents the conceptual framework underlying this study, a description and justification for the selection of the study catchments, as well as the change detection approaches used to assess changes in ecosystem services over time. The second section presents the data collection and processing procedures followed. Section 3 discusses data uncertainties and limitations and explains accuracy assessment procedures undertaken to verify and validate the maps and data produced in this study.

Chapter 4: Results – This chapter presents the findings from this study in line with the posed research questions. It first presents results on habitat changes observed in the study catchments between the two dates used in this study, including changes in their areal extent, changes in spatial location and patterns of habitats. The second section presents the observed changes in ecosystem services in the study catchments, presenting the ecosystem service maps for the study catchments from the two time points, as well as relative assessment of changes in ecosystem service delivery levels. The last section is a synthesis of the findings from this study, discussing the influence of spatio-temporal habitat changes on ecosystem service delivery in the study catchments.

Chapter 5: Discussion – this chapter identifies and discusses the major factors that have influenced habitat and ecosystem service changes in the study catchments. It also discuss implications of findings from this study in catchment management. Also included is a discussion on contemporary issues in mapping ecosystem services which should be taken into account in interpreting conclusions from this study.

Chapter 6: Conclusion and Recommendations – This final chapter presents key conclusions from this study. Also included is a synthesis of the main outcomes from each of the thesis chapters as well as recommendations for both policy and future research.

2 Literature Review

2.1 Chapter introduction

In view of increasing focus and interest towards understanding the ecosystem services concept (Braat, 2012), this chapter set out to review literature on general understanding on the ecosystem services concept, including its origins, definition, classification, any debates, and assessment approaches developed while also identifying key knowledge gaps. Key knowledge gaps identified include:

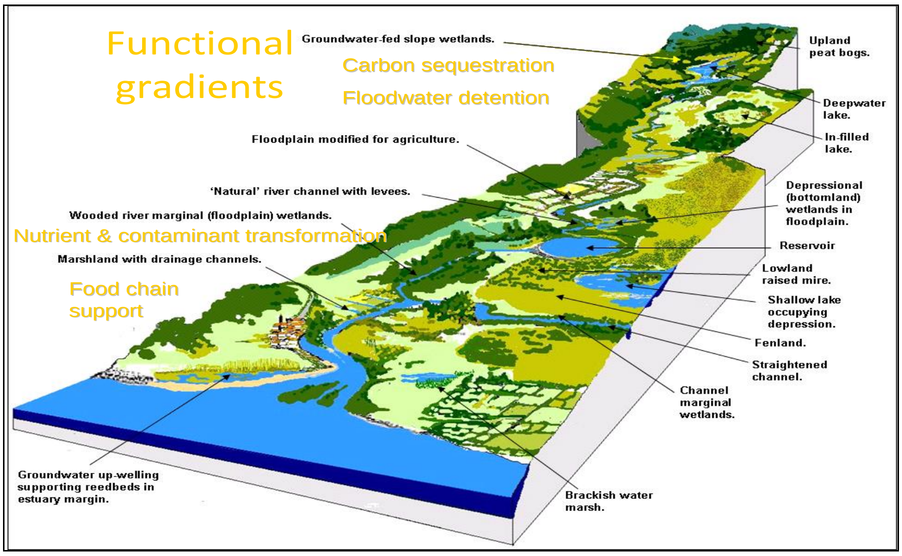
1. the need to describe changes in ecosystem services delivery resulting from habitat/land cover modifications and changes, as there is lack of empirical information on how ecosystem services are changing over time (Haines-Young et al., 2012);
2. the need for more ecosystem service maps to contribute towards establishing the ecosystem service data base (Burkhard et al., 2012), to inform decision making, for involvement of relevant stakeholders and help move the ecosystem services concept into practice;
3. the need to map ecosystem services at local scales such as catchment levels as these are appropriate units for environmental decision making and policy implementation (Kandziora et al., 2013); and
4. the need for more case studies on how to manage river catchments through the ecosystem services approach (UK National Ecosystem Assessment, 2011).

While there are clearly many potential areas of research, the need to understand spatio-temporal changes in ecosystem services resulting from past habitat changes was identified as a key knowledge gap as this has so far, received little attention in the research literature (Haines-Young et al., 2012).

2.2 Section 1

2.2.1 River catchments as important units for environmental management

A River catchment[1] is the geographical area of land that contains a river system, its tributaries and associated ground waters (Ferrier and Jenkins, 2010). Although catchment sizes and scales vary between and within countries, they are all characterised by presence of water moving from river sources through catchment landscapes before discharging into the sea. As water moves through a catchment, it replenishes and supports human life, plant and animal communities and helps maintain healthy ecosystems. The figure below is an extract from the UK-National Ecosystem Assessment (2011), illustrating typical catchment landscapes in the UK, and some of the many biophysical and socio-economic activities and interactions going on within it.



Figure[2] 2-1: Typical catchment landscape in the UK
Source: UK-NEA (2011)

As shown in figure 2-1, there are different types of habitats found within catchments, including upland heath and moorland, freshwater, different types of wetlands, lochs, woodland plantations, enclosed farmland as well as built up areas (UK NEA, 2011). In the UK enclosed farmland is the most extensive habitat type, dominating the catchment landscapes in most of the British countryside (Firbank et al., 2013). Figure 2-1 also shows that catchments are multifunctional and integrated landscapes. Each catchment presents

[1] These are also referred to as watershed or river basins
[2] All figures in this thesis are author`s unless otherwise stated

an array of habitat types and different land use decisions, some of which are connected with both proximity and distance to the river network itself.

Most river catchments; especially in the developed world e.g. Australia, USA, Canada and the UK, have for centuries, been historically engineered and drained through various human activities. Such modifications have resulted in the alteration of ecological integrity of these ecosystems (Tané, 1996), impacting on their regulation processes and biodiversity loss. Despite such a long history of change, Newson (1997), notes that the period of substantial change in river regulation functions was the mid-20th century, marked by the onset of major industrial developments and agriculture mechanisation. The figure below illustrates how various human activities have impacted on the regulation functions of river catchments.

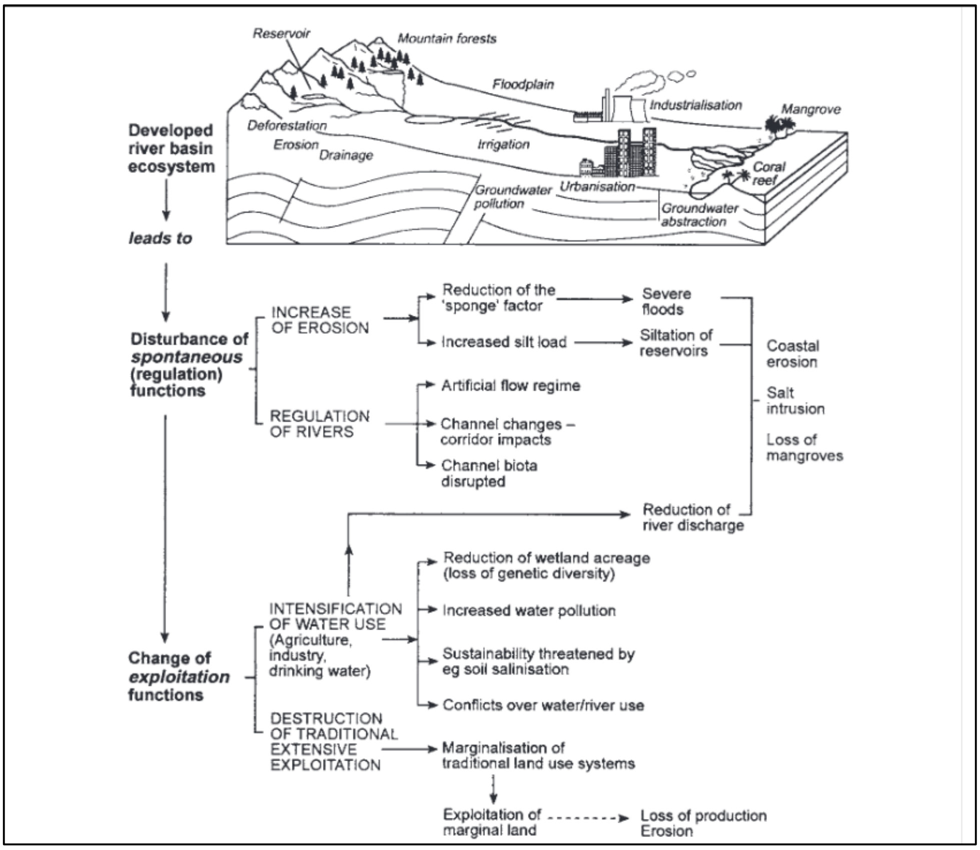


Figure 2-2: Impact of human activities on catchment regulation processes
Source: Newson (1997), developed from Marchand and Toornstra (1986)

As illustrated in figure 2-2, maximisation of tangible benefits through agricultural production, industrialisation and water abstraction impacts on regulation processes such as the alteration of the flow regime, increased soil erosion, loss of biodiversity, increased water and increased flood risk. This further shows that major land uses have an impact on water; altering its quantity, flow rate and quality, as it moves through catchments. This means the impact of such extensive land uses on other catchment processes need to be understood in order to sustainably manage river catchments.

Such a link between river systems, adjacent land uses and habitat types reflects interdependency and interconnectivity of ecosystem processes and biophysical features within river catchments (Ferrier and Jenkins, 2010). This highlights the need for systems thinking and approaches that recognise and integrate these in catchment management (Tané, 1996). In this way, catchments can be viewed as sources, pathways and receptors. For example, diffuse pollution from agriculture moves through runoff pathways into receptor watercourses resulting in water pollution. Similarly, downstream flooding can be exacerbated by upstream land use which might promote runoff into watercourses; which if straightened, can serve as a pathway to increased downstream flooding. Such a conceptualisation can assist in the identification of pressures within catchments and possible solutions towards addressing these (Scottish Environment Protection Agency, 2015a, Werritty et al., 2010).

In recognition of this, the importance of catchments as appropriate natural units for environmental management is widely acknowledged (Tané, 1996, Newson, 1997, Global Water Partnership, 2000a, Savenije and Van der Zaag, 2008, Rieu-Clarke et al., 2015) as they provide a spatial unit for potential integration of all users. Formally, the importance of catchments as appropriate units for management is noted in one of the Dublin principles, which state that, "effective management links land and water uses across the entire catchment". Dublin principles were the outcome from the Dublin International Conference on Water and Environment in 1992 (Global Water Partnership, 2000a). On this basis, Integrated Water Resources Management (IWRM), was proposed as an international policy goal for the management of river catchments during the 1992 UN Conference on Environment and Development in Rio de Janerio (van der Zaag and Savenije, 1999). Biswas (2004), however, notes that IWRM has been in existence for almost half a century and was only rediscovered and vigorously promoted in the 1990s.

2.2.2 Integrated Water Resources Management Approach (IWRM)

IWRM is defined by the Global Water Partnership (2000b) as *"a process that promotes the co-ordinated development and management of water, land and related resources, in order to maximise the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems."* IWRM advocates for the integration of social and physical dimensions of water and a holistic approach to water resources management to ensure their sustainability (Banuri, 2009). The IWRM concept proposed a move from the previously fragmented sectoral approach to catchment management to an approach that integrates all the different water uses (Hendry, 2008). Water users such as agriculture, industries and domestic supply had to ensure that water remains in good quantity and quality to meet the needs of other users including ecological needs of freshwater dependent species (Global Water Partnership, 2000a).

In this regard, river basin organisations established in most river basins in the world recognise the importance of catchments as units for management (Schulze and Schmeier, 2012). For example, countries within the European Union (EU) followed the IWRM approach to frame their water resources management legislation and policies (Savenije and Van der Zaag, 2008). In the EU, the management of catchments has been influenced by the Water Framework Directive (2000). The directive is noted to have come at a time when there were increasing flooding incidences in the EU and water quality was significantly deteriorating (European Commission, 2000). This directive is aimed at improving and protecting surface waters and ground water to maintain good water quality and ecological status (Junier and Mostert, 2012). The WFD requires all member states to ensure that their waters achieve good ecological status by 2015.

This directive also required member states to implement sustainable water resource management at the river basin or catchment scale to ensure integration of land use and water resources management (European Commission, 2000). Prior to this, in Scotland as in other countries in the UK, management of catchments had separate structures, institutions and policies which focussed on different areas such as flooding, water quality, fisheries without a systematic provision for river basin management and planning (Hendry, 2008). The Water Environment and Water Services (WEWS) (Scotland) Act (2003) transposed the requirements of the WFD into Scottish Law and introduced a new framework to manage rivers, lochs, wetlands, estuaries, groundwater and adjacent coastal waters (Hendry, 2008, Spray et al., 2010), using catchments as units for planning and management of the water environment. The use of catchments as units for management was adopted by SEPA in addressing the aims of the WFD and this assisted in focusing issues like diffuse pollution control measures into respective areas.

Such a framework provided a means for addressing environmental challenges related to flooding, diffuse pollution, abstraction and impoundment and engineered alterations to the river beds, banks and shores and pressures in the catchments. It also encouraged engagement of relevant stakeholders like land owners and land managers (Collins et al., 2007, Spray and Comins, 2011) in integrated catchment management.

2.2.3 Implementation success of the IWRM approach critiqued

IWRM has been criticized for lack of measurable positive practical impact, with authors such as Biswas (2004) concluding that IWRM has been a subject of academic debate with limited real world positive practical applications. What Biswas saw was its vague and operationally unusable definition which for example does not provide clarity on how to measure its impact. Biswas (2004, 2008) asks what is meant by integration, who is supposed to promote this concept, what is meant by related resources, what parameters are to be maximised and how can economic and social welfare be determined. As a consequence he further argues that this concept has been interpreted differently by different people turning it into a "*catch-all"* concept.

Cook and Spray (2012), divided criticisms related to IWRM into those concerning knowledge, society and governance. They note that IWRM is criticised for being dominated by engineering and physical sciences. In the process it gave little room for integrating knowledge with other disciplines like social sciences, ecology etc. yet these are equally essential in managing the multi-dimensional nature of catchments.

The societal shortcomings of IWRM are related to its failure to take into account society`s influence on water resources management (McDonnell, 2008). IWRM is said to have given emphasis and much focus on the physical environment as the major factor influencing upstream-downstream relations in the management of shared water resources, with little recognition of the influence of social and political relations (Nhapi et al., 2005). This is particularly so in shared transboundary river basins where social and political relations among riparian states can either impede or promote how they manage their shared water resources (Rieu-Clarke et al., 2015).

In terms of governance, IWRM advocates for public participation and involvement of all relevant stakeholders. It also recognises the need for participation of local communities especially women in the management of water resources. However, practically, this concept is critiqued for not having achieved this (Cook and Spray, 2012).

Similarly, the implementation of the WFD; grounded on this approach, is also reported to have been problematic in the EU (Blackstock et al., 2015). Such problems are related to scientific understanding of relationships between ecosystem status and the impact of measures including complexities and uncertainties associated with these (Hering et al., 2010). Other cited challenges are related to measuring and assessing good ecological status of waters using a scale that's applicable across Europe (Borja, 2005). Also, practically addressing pressures such as diffuse pollution from agriculture is reported to have been a challenge to member states and they instead opted to address point sources of pollution first (Niasse and Cherlet, 2015).

In addition, effective stakeholder engagement in the implementation of the WFD has been questioned (Niasse and Cherlet, 2015, Blackstock and Richards, 2007). The implementation of this directive is argued to have been dominated by a top down approach, reflecting the needs of the central government and legislative requirements, with limited effective engagement of local stakeholder communities (Spray et al., 2010, Spray and Comins, 2011). These authors further contest that, even though structures for local community engagement were set up; through area advisory groups, the process remained top down as these had set agendas and terms of reference controlled by institutions responsible for the delivery of this directive.

In view of these challenges associated with the implementation of the WFD there are increasing considerations on how the ecosystem services concept can assist to deliver the requirements of this directive (Wallis et al., 2011). The ecosystem service concept is considered to be closely similar to IWRM (Cook and Spray, 2012) as they both explore human-environment relations including the complexities with which these are linked (Niasse and Cherlet, 2015). The ecosystem services concept; as shall be discussed in detail in the next section, explores how ecosystems influence human well-being (Millennium Ecosystem Assessment, 2005). As Cook and Spray (2012) put it, "*It recognises the role of nature as the provider for human well-being but also as a victim of pollution and overexploitation by humans.*"

IWRM recognises the interconnectedness of water (upstream and downstream relations) while the ecosystem services concept emphasises human dependency on ecosystems. Both concepts, however, emphasise that competing interests must be integrated in order to implement successful policies (Niasse and Cherlet, 2015). Such competing interests are related to multiple water users and the multiple benefits that catchment ecosystems provide and the need to sustain these.

On the basis of the above, there has been a shift and research into prospects of the ecosystem services concept taking integrated water resources management further to inform the implementation of environmental management directives such as the WFD (Wallis et al., 2011). This is however, not to say the ecosystem service concept is a panacea to such implementation challenges as it has also been hugely contested; as will be discussed later. The following section focusses on the ecosystem services concept describing its origins, how it is defined, how ecosystem services are classified, its key elements and the emerging debates associated with this concept.

2.3 Section 2:

2.3.1 Defining ecosystem services

One of the general and widely accepted definition of Ecosystem Services (ES) comes from the Millennium Ecosystem Assessment (2005) which defined them as "*the benefits people obtain from ecosystems*." The foundation to this definition is reported to have been from two underlying ES definitions by Daily (1997) and Constanza et al. (1997). Daily (1997) defined Ecosystem Services as "*conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life. They maintain biodiversity and the production of ecosystem goods, such as seafood, forage timber, biomass fuels, natural fiber, and many pharmaceuticals, industrial products, and their precursors"*. Constanza et al (1997) suggested that "*Ecosystem goods (such as food) and services (such as waste assimilation) represent the benefits human populations derive, directly or indirectly, from ecosystem functions.*" Even though the above definitions distinguish goods and services, the term ecosystem services is now often used to refer to both goods and services. Since the time when these mostly cited definitions were put forward, a number of definitions have been suggested by different authors such as Boyd and Banzhaf (2007) and Fisher et al. (2009). Braat and de Groot (2012), note that such definitions have had varying attentions to either the ecological basis or economic use.

Braat and de Groot (2012) outline the origins of this concept as dating back to the 1960s and 1970s when environmental pollution, resource scarcity and degradation issues led to increased awareness and scientific based policy development. They further note that the term ecosystem services was developed as a bridging concept between natural and social sciences in the 1980s in response to increased demand to manage economic development under sustainable development. Haines-Young and Potschin (2010), note that Paul and Anne Ehrlich coined this term in 1981 while Mooney and Ehrlich are noted to have used the term "environmental services" in 1970 (Salles, 2011). However, Fisher et al. (2008) explain that appreciation and recognition of the importance of ecosystems to humans and the benefits that humans obtain from ecosystems dates as far back as 1864 with George Parkins Marsh`s writings on Man and Nature.

A marked increase in concepts, approaches and studies aimed at showing the linkages and the relationship between economy and the natural environment, attempts to define this concept and attempts to estimate the economic value of ecosystem services was realised in the 1990s (Braat and de Groot, 2012). This is evidenced by the number of publications that made reference to this concept since the onset of this period. Fisher et al. (2008) found that by 2007, the number of papers that had been published focussing on ecosystem services were approximately 1165 while Potschin and Haines-Young (2011) estimated that between 1966 and 2010, over 5000 articles had been published with 60% of these published since 2006.

Reflecting upon the many ways in which the concept of ES has been defined, Braat and de Groot (2012), Maes et al. (2012a), Fisher et al. (2009) and Nahlik et al. (2012) concluded that there is no agreed definition of this concept. Consequently, Nahlik et al. (2012), offer concerns that the absence of an agreed definition to this concept has led to confusion on what exactly ecosystem services are. These authors further state that the numerous definitions that have been put forward have presented this concept as a "*catch-all*" phrase that refers to anything ranging from, to and within an ecosystem that is beneficial to any living thing. Many commentators (Hauck et al., 2013a, Fisher et al., 2008, Haines-Young and Potschin, 2007), however, acknowledge that these numerous definitions point to the fact that ES are linked, interdependent and form the basis to human life. As Braat and de Groot (2012) put it, "ecosystem services are therefore actually conceptualisations (labels) of the "useful things" ecosystems "do" for people, directly and indirectly".

2.3.2 Classification of ecosystem services

The widely applied interpretation of ecosystem services came from the Millennium Ecosystem Assessment (2005). The Millennium Ecosystem Assessment (MEA) was initiated by the United Nations between 2001 and 2005 with an aim of assessing the state of the world`s ecosystems, identifying the factors that were leading to their degradation and the implications for humanity (Millennium Ecosystem Assessment, 2005). It culminated in the compilation of a synthesis report which has been extensively used as the reference to the ES concept. The MEA classified ES into four broad categories illustrated in the table below:

Table 2-1: Classification of ecosystem services according to the MEA categories

Provisioning Services	Regulating Services	Cultural Services
These are products obtained from ecosystems	These are benefits from regulation of ecosystem processes	These are non-material benefits from ecosystems
• Food e.g. from agriculture or from the wild e.g. fruits • Freshwater • Bio chemicals, natural medicines e.g. from wild harvested species • Genetic resources e.g. for biotechnology, breeding using plant and animal breeds • Ornamental e.g. flowers, shells, sand, gravel • Fibre e.g. wood, timber, wool, roofing thatch	• Climate regulation e.g. regulation of greenhouse gases • Pests and Diseases regulation • Pollination • Water purification • Flood hazard regulation e.g. interception, infiltration, wetlands trap flood waters • Water quality regulation • Erosion hazard regulation e.g. regulation of erosive effect of water flow • Waste regulation • Air quality regulation e.g. trapping particulate matter • Soil quality regulation	• Spiritual and religious • Recreation and tourism • Aesthetic experience and values • Inspiration • Educational • Sense of place • Cultural heritage
Supporting services		
These are services necessary for the production of all other ecosystem services		
Soil formation	Nutrient cycling	Primary production

Source: MEA (2005)

The MEA categorisation shows that ecosystems provide services that range from tangible benefits such as food to intangible benefits like aesthetic beauty, including a range of regulatory services such as water purification and climate regulation through e.g. carbon storage. They also provide supporting services e.g. nutrient cycling which are considered to be necessary for the production of the other three ES categories as these are not directly used by humans. However, other scholars e.g. Blackwell and Pilgrim (2011) exclude supporting services because their interpretation is argued to overlap with other categories and can lead to double counting especially in cases that involve economic valuation of ecosystem services.

Despite its wide use, the MEA classification of ecosystem services has been criticised for confusion among different ES categories especially as some supporting services can also be classified otherwise e.g. biodiversity can be a supporting service; underpinning some ecosystem services but can be a cultural ecosystem service e.g. wildlife appreciation, scenic places, spiritual and other recreational values (Mace et al., 2012). Such a confusion can result in double counting of such ecosystem services in valuation attempts (Boyd and Banzhaf, 2007). Wallace (2007) criticises the MEA classification for mixing processes (means) for achieving services with the services themselves (end). He for example argues that supporting and regulating services are means of achieving provisioning and cultural services e.g. water regulation is a means to achieve potable water. This is seconded by Fisher et al. (2008) who argues that services such as supporting can also be regarded as regulating services and such confusion could pose a challenge to effective decision making.

Consequently, other classification systems which have refined the MEA classification further have been proposed, The Economics of Ecosystems and Biodiversity (TEEB) and Common International Classification of Ecosystem Services (CICES). TEEB was launched in 2007 and it focusses on economic costs of biodiversity loss and ecosystem degradation (TEEB, 2010). It mainly follows the MEA classification of provisioning services, regulating, habitat services, cultural and amenity services but excludes supporting services.

CICES closely matches with TEEB ES categories and focusses on the final outputs/products from ecosystems i.e. things that are directly consumed, used or enjoyed by people (Science for Environment Policy, 2015). It includes provisioning, regulating, maintenance and cultural ES and excludes the supporting ES. It differs from TEEB in the treatment of the habitat services. This framework is noted to be useful in the integration of values of ecosystems in accounting frameworks and avoiding double counting in ecosystem services (Maes et al., 2013). In this regard, it has been proposed as a standard typology in mapping and assessment of ES by those studies that seek to integrate ES mapping, environmental accounting and economic valuation. Both the TEEB and CICES consider supporting ecosystem services as part of the ecosystem processes and functions.

There are also country level classification systems that have been used but they closely follow the MEA classification. For example, classification of ecosystem services in the United Kingdom National Ecosystem Assessment (UK NEA), broadly followed the MEA categories of provisioning, regulating, cultural and supporting. However, the UK NEA went further to distinguish whether such ecosystem services were final or intermediate processes to allow for valuation of final ecosystem services as illustrated in the table below (UK-National Ecosystem Assesment, 2011).

Table 2-2: Classification of ecosystem services in the UK National Ecosystem Assessment

Ecosystem processes/intermediate services		**Final ecosystem services**	
Supporting services	-Primary production -Soil formation -Nutrient cycling -Water Cycling	**Provisioning services**	-Crops, livestock, fish -Trees, vegetation, peat -Water supply -Wild species diversity
-Decomposition -Weathering -Climate regulation -Pollination -Disease and pest regulation -Ecological interactions -Evolutionary processes -Wild species diversity		**Cultural services**	-Wild species diversity (recreation) -Environmental settings (recreation, tourism, spiritual/religious)
		Regulating services	-Climate regulation -Pollination -Detoxification and purification in soils, air and water (pollution control) -Hazard regulation (erosion control, flood control) -Noise regulation (noise control) -Disease and pest regulation (disease and pest control)

Source: UK-NEA (2011)

As shown in table 2-2, provisioning and cultural ecosystem services were classified as final services while regulating services were either final or intermediate services or processes and supporting services were either intermediate services or processes. The table also shows that some ecosystem services appear in more than one category such as wild species diversity which can be a cultural ecosystem service but also a supporting ecosystem service.

To give a brief background, the UK NEA was undertaken following, the MEA`s recommendation for countries to assess the state of their ecosystems. The UK National Ecosystem Assessment (2011) took the MEA approach further and in much detail with a focus on the individual countries within the UK, providing a baseline assessment of the benefits that the natural environment provides to society and its contribution to the national wealth. The UK-NEA also gave a state level detailed account of ecosystem services, their trends, drivers, scenarios about possible futures and policy options that might lead to the development or avoidance of these future scenarios.

2.3.3 The link between ecosystems and human well being

The Millennium Ecosystem Assessment (2005), also illustrated the link between ecosystems and human well-being. The figure below shows how the MEA ES categories relate to constituents of human well-being.

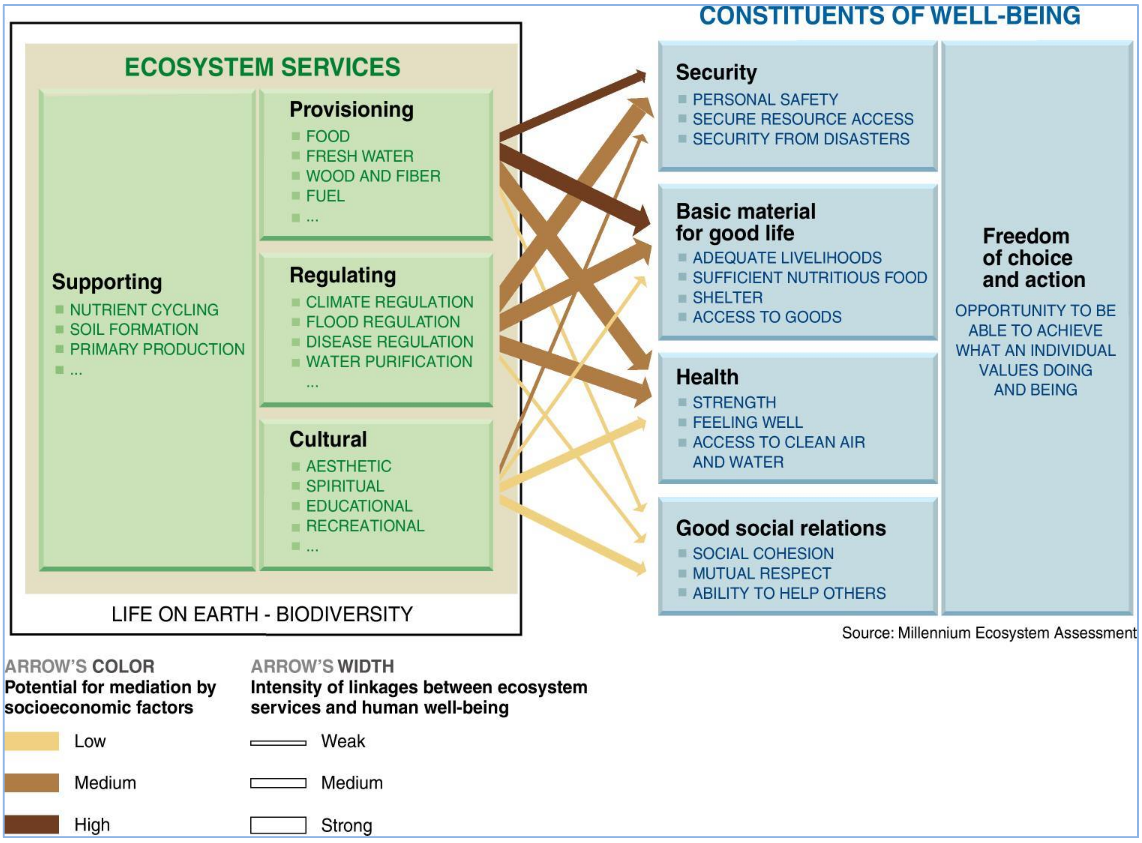


Figure 2-3: Link between Ecosystem Services and Human well-being

Source: Millennium Ecosystem Assessment (2005)

As illustrated in the diagram above, the width of the arrows show the relative importance of each of the ES categories in contributing to different constituents of human well-being. For example, there is a strong link between provisioning services such as food and their contribution to basic materials for good life and health, such as access to sufficient nutritious food. The figure also shows that ES enhance multiple components of human well-being, emphasizing the dependency of humans on nature.

The Millennium Ecosystem Assessment illustration on how ecosystem services contribute to human well-being led to what probably can be viewed as a new way of thinking (paradigm): the Ecosystem Services Approach (Seppelt et al., 2011, Wallis et al., 2011, Waylen et al., 2014). This approach focusses on understanding the services that the natural environment provides to humans and then managing the environment so that the provision of these services is sustained over long term (Millennium Ecosystem Assessment, 2005, Wallis et al., 2011). The key elements of such an approach, as identified by Turner and Daily (2007) include:

- Identifying multi benefits offered by ecosystems including location of such benefits, identify the ecosystem services they offer, understand the state of these ecosystems including historic influences.
- Valuing the multi-benefits from ecosystems and analysing how they contribute to human well-being. Engaging with all relevant stakeholders on prioritised ecosystem services, options analysis, opportunities analysis and trade off analysis.
- Informing policies and decision making on how the society values and perceive ecosystem services provided in a locality.

2.3.3.1 Ecosystem Approach vs Ecosystem Service Approach?

The Ecosystem Approach, is defined by the Convention on Biological Diversity (CBD) as "*an integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way*". The Ecosystem Approach (EA) was adopted as the primary framework for action by the signatories to the CDB in 1995. Its overall aim is to halt the rate at which biological diversity is being degraded and lost and its main objectives are:

- Conservation of biological diversity
- Sustainable use of the components of biological diversity
- Fair and equitable sharing of the benefits arising out of the utilisation of genetic resources.

The EA is based on 12 principles (Waylen et al., 2014) (refer to appendix 2). These principles can be summarised as relating to the management of ecosystems, need for evidence-based science, involvement of stakeholders and considering societal choices. These principles essentially advocate for an integrated approach to management while also promoting an adaptive and flexible approach to the management of natural resources, reflecting uncertainty and the unpredictable nature of ecosystems and how they respond to change (DEFRA, 2010). The principles are interrelated, linked and emphasize the need for a holistic approach (Waylen et al., 2014).

The Ecosystem Services Approach is intricately linked to the Ecosystem Approach and it further builds on the Ecosystem Approach. Its difference to the Ecosystem Approach is that, it goes further and gives a conceptual framework for the assessment and management of ecosystems. Its conceptual framework shows how ecosystem services and values flow from ecosystems to humans and how this contributes to human wellbeing, an aspect which does not feature in the Ecosystem Approach. The ESA could be viewed as the lenses through which the wider Ecosystem Approach can be implemented. In fact, ecosystem services are mentioned as one of the EA principles (i.e. Principle number five), which states that, "conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach".

2.3.4 Conceptual frameworks for understanding ecosystem services

On the basis of the above, frameworks to provide theoretical understanding of ecosystem services have recently been developed. One such notable example is the ecosystem service cascade (Figure 2-4) after Haines-Young and Potschin (2010). The ecosystem services cascade; as shown in the figure, is a stepwise illustration of the connection between ecosystems and human well-being. This has also been adopted as a common conceptual framework for the assessment, mapping and valuing; especially in non-market terms, of ecosystem services (Maes et al., 2012a).

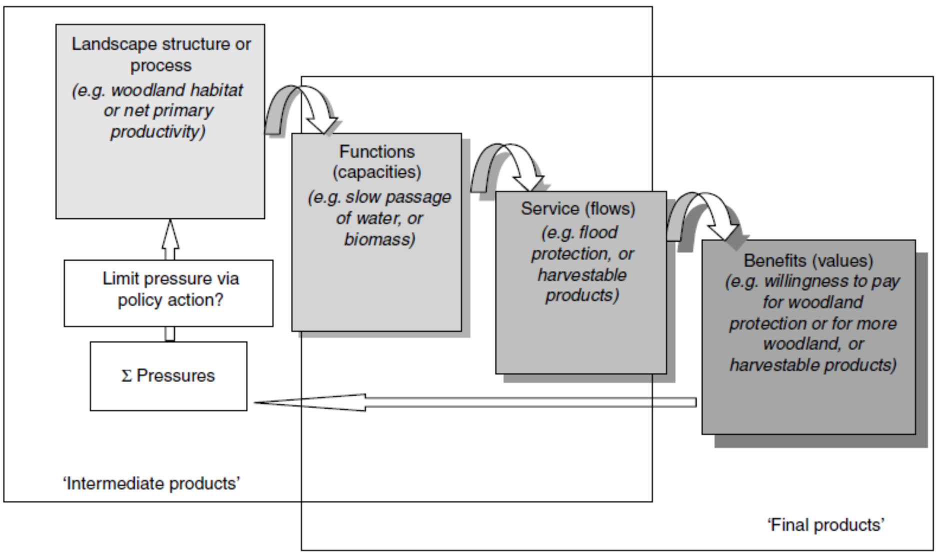


Figure 2-4: Ecosystem Service Cascade

Source: Haines-Young and Potschin (2010)

As illustrated in figure 2-4, ES are derived from ecosystem functions which are underpinned by biophysical structures and processes. In this case, an ecosystem function refers to the capacity or potential to deliver services while ecosystems provide the necessary structure and processes that underpin ecosystem functions. Ecosystem services are derived from functions and the benefits of such services are determined by the values that people place on these. Haines-Young and Potschin (2010), explain that this can be viewed as a cascade linking the ends of a production chain, with elements of ecological and biophysical structures and processes on one hand and elements of human well-being on the other hand.

The ES cascade diagram is, however, criticized for showing that the transition from ecosystem services to benefits appears to be a simple step, yet in reality this is a complex process as people have different appreciation, interests and perceptions (Braat and de Groot, 2012). The ecosystem service cascade has also been interpreted to show that ecosystem services flow in one direction from ecosystems to human well-being without any feedback or input from the receiving box. It has thus been adapted and modified to reflect these feedbacks and to separate benefits from values. TEEB (2010) for example, modified the ES cascade to show the feedback between ecosystems, ecosystem services and human well-being as illustrated in the figure 2-5.

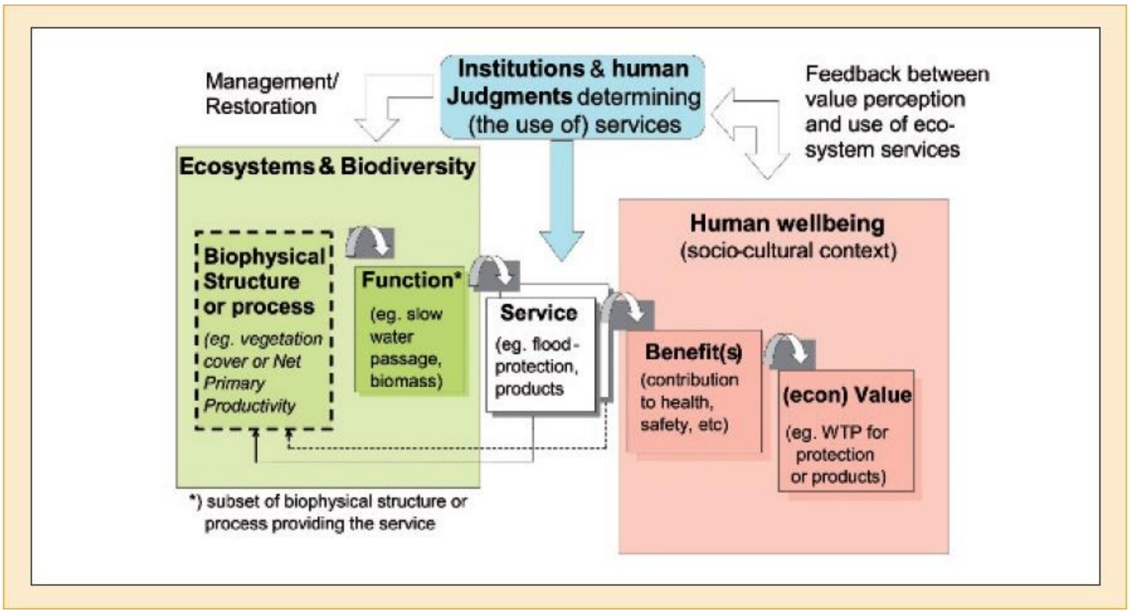


Figure 2-5: The Economics of Ecosystems and Biodiversity pathway diagram

Source: TEEB (2010)

As shown in figure 2-5, TEEB (2010) places ecosystem services between natural and human systems. It identifies benefits for people flowing from services delivered by ecosystems and separates benefits and values. TEEB (2010) also added positive feedbacks through institutions, judgements, management and restoration which connects social sciences with natural sciences.

Other conceptual frameworks based on the ecosystem service cascade have also been proposed by e.g. Maes et al. (2013) (figure 2-6). This was particularly developed in line with action 5 of the EU Biodiversity Strategy to 2020, aimed at mapping and assessing ecosystems and their services in the EU member states.

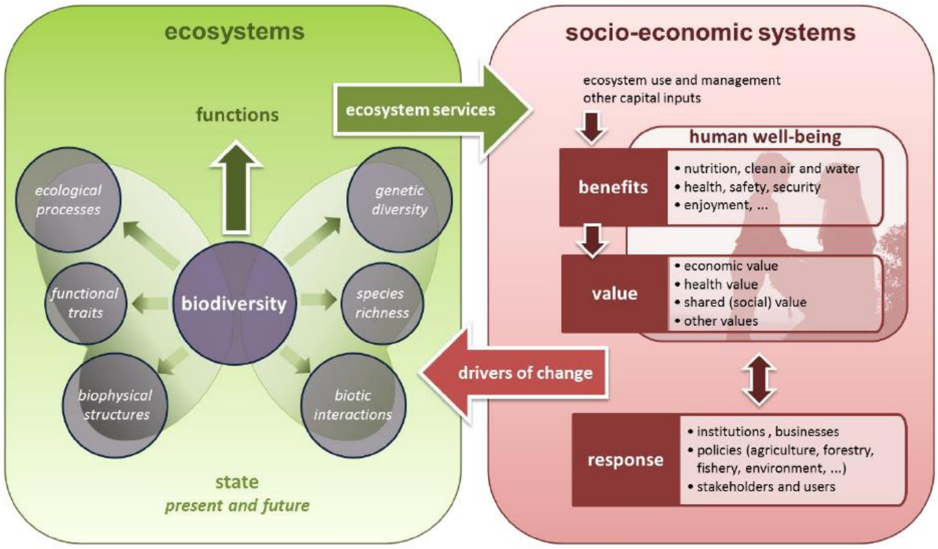


Figure 2-6: Proposed conceptual framework for EU wide ecosystem service assessments

Source: Maes et al. (2013)

As illustrated in the figure above, this conceptual framework depicts elements of both the ecosystem services cascade and the Driver, Pressure, State and Impact framework (Maes et al., 2013). Ecosystem services link ecosystems and socio-economic systems. Ecosystems services e.g. clean water in turn influence and enhance human well-being. Beneficiaries to such ecosystem services include stakeholders and institutions which in turn impact on ecosystems through direct or indirect drivers of change. Such drivers of change for example could be policies aimed at achieving desired future state of ecosystems. Ecosystems on the other hand, are shaped by interactions of living organisms and abiotic environment of which biodiversity plays a key role in the structure of ecosystems, essential for maintaining ecosystem processes and functions (Mace et al., 2012).

There are also other conceptual frameworks that have been developed to understand ecosystem services. For example, Rounsevell et al. (2010) modified the Drivers-Pressures-State-Impact-Response (DPSIR) framework into a Framework for Ecosystem Service provision (FESP to inform assessments of environmental change drivers on ecosystem service provision. Fisher et al. (2013) also provide a review of other conceptual frameworks that have been used to understand the link between poverty and ecosystem services.

2.3.5 Ecosystem services as a contested concept

The emergence of the ecosystem services concept evoked a huge area of debate between opponents and proponents of this concept. One of the central areas for debate focusses on attempts at economic valuation of ES. This and associated concerns on commodification of nature is probably the most contested aspect of the ES concept (Engel and Schaefer, 2013, Braat and de Groot, 2012, Chan et al., 2012, Salles, 2011, Spangenberg and Settele, 2010). Opponents to economic valuation argue that ecosystems provide a range of services, but that economic valuation does not capture all aspects of services, as it does not encompass all dimensions of value (Chan et al., 2012). Salles (2011), following Spangenberg and Settele (2010) and Kumar and Kumar (2008), argue that an economic perspective to valuation is anthropocentric referring to individual`s preferences and individual utility maximisation. These authors further argue that such a perspective excludes intrinsic values of nature and might not be suitable for some ecosystem services like cultural identity which are not only dependent on individual preferences but are also influenced by prevailing social and cultural practices.

Economic valuation methods are asserted to be an underestimation or underrepresentation of the actual value of ecosystem services as they are fraught with inconsistencies (Fisher et al., 2009, Hauck et al., 2013a). They also do not adequately explore less tangible social issues and intrinsic values related to, for example cultural services (Chan et al., 2012). On the other hand, proposed economic methods for the valuation of non-marketed ecosystem services like cultural services are beset with uncertainties and are argued not to reflect the complexities associated with these (Salles, 2011, Satterfield et al., 2013, Spangenberg and Settele, 2010). For example, approaches like Willingness to Pay (WTP) and

Willingness to Accept (WTA), which are both stated preference (contingent valuation) measures to economic valuation of non-marketed ecosystem services, can yield different results when applied at the same place (Spangenberg and Settele, 2010). This is seconded by Salles (2011) who also observed that approaches such as WTP are difficult to apply among people who are poorly informed about biodiversity, and willingness to pay increases with the level of information provided.

Proponents of economic valuation (de la Hera et al., 2011, Fisher et al., 2008, Verma and Negandhi, 2011) argue that ignoring economic analysis will in a way be reducing the weight of arguments that can inform decision making for the management of the natural environment, as most planning decisions are based on such economic criteria. Spangenberg and Settele (2010) like Tengberg et al. (2012), however, counter this argument and assert that policy relevant recommendations can still be made based on qualitative assessments, including other non-economic approaches to valuing ecosystem services. Such approaches involve qualitative (e.g. for cultural services – expert scoring checklist), quantitative (e.g. for amount of nitrogen in a water body – use of biophysical characteristics), geospatial mapping (linking quantitative data to geographical information) and economic valuation (Russi et al., 2013, Wallis et al., 2011, Turner et al., 2008). This is supported by TEEB (2010), which acknowledges that values of ecosystem services can be measured through various methodologies; some of which capture the intrinsic value of ecosystem services, without necessarily using monetary values or economic valuation.

A second critique to the ecosystem services concept relates to the absence of an agreed, clear and consistent definition and classification of ecosystem services. This is argued to have led to a lag in the implementation of this concept "the implementation gap" (Nahlik et al., 2012). In response to this, other commentators (Jackson et al., 2013a, Hauck et al., 2013a) propose that the definition by Boyd and Banzhaf (2007) *"ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being,"* should be adopted as the agreed definition for this concept as it is suitable for measuring, valuing and communicating ecosystem services. Likewise, other commentators (Chadwick J et al., 2005, Haines-Young and Potschin, 2007, Maes et al., 2013) suggest that an agreed classification system will also assist in avoiding double counting. As such they propose that for valuation purposes a classification system such as CICES should be adopted.

Other authors including Fisher et al. (2009) caution towards the adoption of a single classification system given the complexity and dynamic nature of ecosystems and the way these are perceived by different people in different contexts and places. Costanza (2008) regards the absence of an agreed definition and classification of ecosystem services as an indication of the development of this concept which could lead to its improvement through invoking more ideas. He further argues that the definition and classification of ecosystem services adopted depends on the intention of the assessment.

Thirdly, critics argue that the ES concept conflicts with biodiversity conservation goals (Schröter et al., 2014), as there is lack of empirical evidence on the relationship between these, related to scientific uncertainty non-linear relationships and tipping points. There are also questions on whether the adoption of this concept can protect biodiversity (Science for Environment Policy, 2015). Conversely, proponents argue that ecosystem services and biodiversity largely overlap and the ecosystem services concept compliments biodiversity conservation goals (Mace et al., 2012). Biodiversity underlies fundamental ecosystem processes; playing a key role in ecosystem service provision, and it can also be an ecosystem service e.g. cultural ecosystem services of wildlife, scenic and aesthetic beauty etc. Mace et al. (2012) concluded that considering biodiversity within the context of ecosystem services is an opportunity rather than a threat. This is supported by Leisher (2015), who is of the view that bringing to the fore the multi benefits provided by nature, including the less obvious and less valued regulating and supporting ecosystem services could provide compelling reasons for biodiversity conservation.

The ecosystem services concept is also criticised for its potential to promote exploitative human-nature relationships (Schröter et al., 2014). Counter-arguments to this view point to the fact that such an emphasis on human-nature relationship raises awareness on humanity`s dependency on nature (Vihervaara et al., 2010) especially in the Western World where communities are disconnected from nature (Schröter et al., 2014). This concept can also assist communities to identify a full range of ecosystem services provided by nature (Brauman et al., 2014) and help them recognise the value of such services (Haines-Young and Potschin, 2007, Schröter et al., 2014). Thereby offering an easier way to communicate with stakeholders and policy makers.

Norgaard (2010) notes that, the ES concept started as an eye-opening metaphor in communicating and raising awareness on the importance of nature to humanity but has been transformed into a complexity blinder, as it is currently viewed as the ultimate scientific model that can inform policy development and address ongoing environmental challenges. Norgaard (2010) argues that the ecosystem services concept is too simplistic to guide policy development and insufficient to address ecological, economic, and political complexities related to current and future environmental challenges, as some of these challenges require major institutional changes. Instead, he views the ecosystem service concept as part of a larger solution to environmental challenges.

Despite this contention around the ES concept, there is optimism that this concept can be used to address current environmental challenges (Schröter et al., 2014). The ES concept is gradually being used as an indication of policy or management attention on sustainable management of ecosystems and the services they provide to humanity (Hauck et al., 2013a, Plant and Ryan, 2013, Tengberg et al., 2012). In the UK, there is increasing interest in the adoption of this concept to manage the natural environment (UK-National Ecosystem Assessment, 2014). For example, the UK National Ecosystem Assessment (2011), recommended the need for more case studies in order to understand how river catchments can be sustainably managed through the ecosystem services concept.

Similarly, in the EU, the ecosystem service concept is increasingly being tailored into environmental policy landscape (Maes et al., 2012a) as evidenced by, for example, the inclusion of this concept in the EU Biodiversity Strategy for 2020. In particular, Target 2 of this strategy is aimed at maintaining and enhancing ecosystems and their services by 2020 through establishing green infrastructure and restoring at least 15% of degraded ecosystems (European Commission, 2011). Also in the EU, the ecosystem service concept; as discussed in the previous section, is expected to assist towards the implementation of the Water Framework Directive (Wallis et al., 2011, Blackstock et al., 2015). In Scotland, the Scottish Government recently completed pilot projects on the possible implementation of the National land use strategy on mitigating the potential impacts of climate change using the ecosystem services concept (Spray, 2014).

Elsewhere, the ecosystem service concept has also been adopted in South East Queensland in Australia as a framework for planning and management of ecosystem service delivery within the catchments in this region (Maynard et al., 2015). The New York Catskill watershed case study in the USA is also another common example of the successful application of the ecosystem services concept in integrated catchment management (Brils et al., 2015).

In this way, the ES paradigm has emerged as a crucial link between science, decision making and policy making (Villa et al., 2009), strengthening the environmental policy-science interface. It is arguably emerging as one potential cornerstone of sustainability science, drawing attention from scientists, policy makers and practitioners while also promoting interdisciplinary research bridging biophysical and social sciences. On this basis, ideas on assessment and management of ecosystems through this approach have gradually been developing (Fish, 2011, Haines-Young and Potschin, 2010). For example, mapping of ecosystem services has been identified as one essential element in the assessment of ecosystem services to inform environmental decision making (Seppelt et al., 2011) while contributing towards moving the ES concept from theory into practice. The next section discusses the importance of mapping ecosystem services and reviews approaches, methods and tools that have been developed to map ecosystem services.

2.4 Section 3

2.4.1 Mapping and Assessment of ecosystem services

In recent years, there has been an upsurge in studies aimed at mapping the spatial distribution of ES (Martínez-Harms and Balvanera, 2012, Cowling et al., 2008, Maes et al., 2012a) which Vigerstol and Aukema (2011) describe as a stage of "many flowers blooming", indicating increased interest in mapping of ecosystem services. This is supported by findings from a study by Crossman et al. (2013) which showed that the number of ES mapping studies increased from one study in 1996 to more than 10 per year since 2008. Despite this seemingly increasing interest in mapping ecosystem services, there are calls for more ecosystem service mapping studies. More ES mapping studies are expected to contribute towards the development of an independent ecosystem services mapping database (Burkhard et al., 2012) as there is currently lack of such a database (Schulp et al., 2014). Moreover, a recent review by Andrew et al. (2015) showed that, mapping ecosystem services is one aspect which is lagging; in terms of number of studies done to this effect, compared to the number of studies focussing on other aspects of this concept such as economic valuation.

As discussed in the previous section, mapping ecosystem services is an integral part of the ecosystem services approach (Turner et al., 2008). A number of reasons justifying the importance of mapping ecosystem services, have been put forward. Daily and Matson (2008) and Seppelt et al. (2011) for example, identified ecosystem services mapping as an essential pathway for improving the relevance and mainstreaming of the ES concept into policy development. This was particularly in view of mounting concerns and criticisms over the lack of practical evidence on the implementation success of this concept.

Ecosystem service maps are deemed useful in a number of ways, including spatial representation of ecosystem service supply areas, their flows and demand areas (Grêt-Regamey et al., 2015), while also demonstrating how these vary across space and time (Brauman et al., 2014). Ecosystem service maps can also show multiple ecosystem service providing areas within a landscape, illustrating spatial patterns, trends, diversity, synergies and trade-offs among different ES (Raudsepp-Hearne et al., 2010, Maes et al., 2013).

By mapping ecosystem services, "*invisible*" ecosystem services are made "*visible*", as both common tangible and less obvious ecosystem services are explicitly spatially presented, otherwise not revealed in simple habitat mapping. This allows for visualisation of ecosystem service supply areas as well as understanding spatial variations in ecosystem service supply within catchments. This can inform decision making and management action on areas of multiple ecosystem service provision (Verhagen et al., 2015). As illustrated in the figure below, by mapping ecosystem services, the contribution of land parcels and habitats to both obvious ecosystem services like water supply and less obvious ecosystem services like soil carbon storage in catchment landscapes can be spatially visualised. Characterisation of such spatial configurations of habitats and their role in ecosystem service delivery can help inform management of landscapes (Syrbe and Walz, 2012).

Figure 2-7: Mapping ecosystem services aids in visualisation of both tangible and less obvious ecosystem services
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By visualising multiple ecosystem service supply areas within landscapes, it is possible to identify areas where there are opportunities for multiple ecosystem provision, as well as areas of conflict in ecosystem service provision. Similarly, degraded or threatened areas that need to be protected, managed or restored for enhanced multiple ecosystem service provision can be identified (Villa et al., 2009, Vigerstol and Aukema, 2011, Maes et al., 2012a). Furthermore, ecosystem service maps could also be used in trade-off analysis and assessing the impact of unintended consequences on other ecosystem services (Spray, 2014).

ES maps are also particularly valuable in aggregating ecological complexities into ways that can be easily conveyed to non-experts (Medcalf et al., 2014, Burkhard et al., 2012, Hauck et al., 2013b). In so doing, ES maps serve as a powerful communication tool (Hauck et al., 2013b). For example, in the Scottish Government Land use pilot project in the Scottish Borders, ES maps were used in local stakeholder consultation workshops (Scottish Borders Council, 2015) to help identify and verify relevant and important ES in their locality. They also used these to prioritise land uses within the Scottish Borders and inform the formulation of opportunity, interaction and multi benefit ES maps. In the process, the impact of policy decisions and possible future scenarios was communicated in ways that local stakeholders understood and could relate to (Spray, 2014). Similarly, findings from the study by Hauck et al. (2013b) showed that maps could serve as a contractual agreement between stakeholders and authorities in voluntary conservation measures like agri-environment schemes, showing agreed sites where such measures could be implemented.

Ecosystem service maps can provide a baseline against which to assess both current and future ecosystem management intentions and policies against set targets (Jiang et al., 2013, Antrop, 2005). Past ES maps for example, can provide a baseline on the state of past ecosystems and assist in identifying areas where management measures can be implemented. For example, in addressing legislation such as the Flood Risk Management Act (2009) and associated policy objectives, ES maps can be used to spatially identify sites within catchments with high flood control potential, requiring either to be protected or restored. The added value of using ecosystem service maps in this instance rather than habitat maps alone would be their ability to identify and show areas within catchments where policies could achieve trade-offs, synergies and multiple benefits (Schaafsma et al., 2015).

However, like any other maps, ES maps are a generalisation of reality; reflecting judgements and systematic bias and prone to similar shortfalls related to maps in general. Literature on critical cartography for example, query the legitimacy of maps, viewing them as depicting existing power relations of map creators as they decide what to reveal and what to conceal (Wood, 1992). Monmonier (1991) for example, notes that in sectors such as Town and Regional Planning, property developers manipulate maps by omitting details that would jeopardise the success of their application.

Hauck et al. (2013b) explain how one of the key informants from their study viewed ES maps as having “an air of authority”. This can be interpreted as an expression of a concern over ES maps seen as instilling existing power relations especially if they are used as legal documents in voluntary conservation schemes. Additionally, there are concerns over showing high ecosystem service supply areas as this could actually lead to the exploitation of such areas rather than protection. On the other hand, degraded areas with low ecosystem supply potential could be stigmatised (Hauck et al., 2013b).

The use of ES maps can introduce biases due to the types of ecosystem services being mapped and provide limited information on other ecosystem services (Nemec and Raudsepp-Hearne, 2013), given that it could be a challenge to spatially represent some ecosystem services like cultural ones. However, these also need to be accounted for in decision making. Furthermore, lack of data at appropriate scales, as is often the case for ecosystem services, can hinder production of maps that are detailed enough to inform decision making. Lack of accurate data to map and validate ecosystem service maps (Willemen et al., 2015) also raises questions about their credibility (Hauck et al., 2013b).

Due to these limitations, Hauck et al. (2013b) and Schulp et al. (2014) caution that ecosystem service maps need to be used with great care and responsibility. These authors also recommend involvement of stakeholders during the mapping of ecosystem services, as this can contribute to the transparency, legitimacy and relevance of the generated ecosystem service maps.

Mapping of ecosystem services, however, remains important as many authors (Egoh et al., 2008, Petter et al., 2013, Vihervaara et al., 2010) support the potential use of ecosystem service maps in informing environmental policy, decision making and environmental management. The importance of ecosystem service maps is currently reflected in biodiversity conservation strategies such as the EU Biodiversity Strategy for 2020. Action 5 of the EU biodiversity strategy entail mapping and assessment of ecosystems and their services in the member states (European Commission, 2011). This reflects the importance of mapping ecosystem services in contributing towards halting biodiversity loss and degradation of ecosystem services. More so, the UK NEA (2011), identified the need to understand changes in ecosystem services as there is lack of empirical information about how these are changing over time. In particular, few studies have attempted to map and assess past ecosystem services (Haines-Young et al., 2012).

2.4.2 Ecosystem service mapping methods and approaches

Different diverse methods and approaches, integrating both spatial and non-spatial data have been used to map ecosystem service supply areas (Andrew et al., 2015). Willemen et al. (2015) attribute such diversity in ES mapping approaches to similar variations in the definition and classification of ecosystem services used by different studies. Andrew et al. (2015) like Martínez-Harms and Balvanera (2012) observed that other studies use a combination of methods to map different ES types, while in some cases the same ecosystem services are mapped using different methods. Martínez-Harms and Balvanera (2012) conducted an extensive review of methods that have been used to map ecosystem services in current practice and concluded that generally such methods can be classified into following five categories.

1. Use of look-up tables- this uses constant relative ES values assigned to each land cover (LC) type. Such values are obtained from previous studies at other places. LC types are assigned values based on their potential to provide a particular ES.
2. Expert knowledge – in which experts are requested to assign and rank environmental variables such as land cover or habitat types, based on their knowledge about the potential of such variables to supply ES.
3. Use of causal relationships – this relies on existing documented knowledge (scientific literature) about the relationship between an environmental variable and the supply of ES.
4. Extrapolation of ES values from primary data – this involves the extrapolation of the ES supply relative value for a given environmental variable e.g. land cover class based on measured field data.
5. Regression models – this entails modelling of the relationship between field samples of ES and readily measurable environmental variables. Such models account for underlying processes that affect ES supply such as soil type, plant species and topography which might influence the delivery of a selected ES.

Eigenbrod et al. (2010b) on the other hand broadly divided the methods for mapping ES into: (1) those that require primary data from within the study area and (2) those based on land cover proxies or proxies based on logical combinations of likely causal variables informed by prior knowledge. More recently, Andrew et al. (2015) divided the ES mapping approaches into: (1) those that involve direct mapping of ecosystem services with survey and census approaches, (2) point-based measurement of ecosystem services using empirical models, and (3) use of rule based models which at times are used alongside extrapolation and data integration to proxy the distribution of ecosystem services.

The above reviews all show that ES mapping is largely based on the use of proxy data or integration of different proxy variables. To substantiate this, a quantitative review by Seppelt et al. (2011) showed that less than 40% of reviewed studies derived their results from primary data. This is because many ES do not have primary data as there is still lack of appropriate data for quantification of ES (Science for Environment Policy, 2015). Databases for mapping ES are currently under development and this justifies the use and reliance on proxy data (Burkhard et al., 2012). Of these, use of LC/LU/habitat based proxy data in some cases integrated with other datasets or aspatial data is noted to be the most common approach in ES mapping (Seppelt et al., 2011, Eigenbrod et al., 2010b, Andrew et al., 2015).

The use of land cover/habitat data as proxy is argued to be an appropriate base for mapping ecosystem services (Burkhard et al., 2009, Sohel et al., 2015). This is because such remotely sensed data is available for many places in the world, at different scales including regions such as Africa where there is otherwise limited data to inform the mapping of ecosystem services (Nemec and Raudsepp-Hearne, 2013, Leh et al., 2013, Andrew et al., 2015, Vrebos et al., 2015a). Even though the level of detail and accuracy of the ES maps generated in data scarce regions might be limited, using such land cover data can provide indications on ecosystem service provision which can be followed by detailed assessments.

In using land cover/habitat data, an ecosystem is considered at the scale of a habitat or land cover type. This is because habitats are one of the primary landscape units providing ecosystem services (Syrbe and Walz, 2012) and their classification significantly overlaps with that of ecosystems (UK-NEA, 2011). Habitats are also used as ecosystem boundaries as they give spatially identifiable and delineable boundaries (Vermaat et al., 2015). For example in the EU, the CORINE land cover classes have been proposed and used by a number of studies to map ecosystem service supply areas (Nedkov and Burkhard, 2012, Burkhard et al., 2012, Maes et al., 2013). Similarly, the UK NEA (2011) mapped ecosystem services based on recognised broad habitat types found in the UK. Other example studies that used LC/LU data to map ecosystem services include: Jiang et al.,

(2013) in Dorset (England); Lautenbach et al., (2011) in Leipzig (Germany); Leh et al. (2013) in Ghana and Cote d`Ivoire; Sohel et al. (2015) in the Lawachara National Park in Bangladesh; Li et al. (2007) in China; and Haines-Young et al. (2012) in the European Union.

The relationship between habitats/LC/LU and ES provision can be understood as a cause-effect one (causal relationship) grounded on the assumption that different habitats/LC equate to different (distinct) ecosystems and they in turn produce different combinations of ecosystem services. Not only that, but that these can be competently and rigorously identified. This means that prior to mapping ecosystem services, the distribution of these needs to be understood. On this basis, changes and alterations to land cover or habitats impacts on ecosystem service delivery (Burkhard et al., 2012, Kandziora et al., 2013, Haines-Young et al., 2012, Millennium Ecosystem Assessment, 2005, UK-NEA, 2011). Linked to this is the assumption that in different types of habitats, ecosystem functions and species necessary for ecosystem service delivery are in existence and functioning as expected. Habitats in a good conservation status are for example considered to be healthy ecosystems with a higher potential to supply regulating and cultural ES (Maes et al., 2013).

2.4.3 Ecosystem services mapping tools

Alongside the different ES mapping approaches, tools for mapping ecosystem services are also diverse although with a similar end goal of spatial representation of ecosystem services. Below is a brief description of the common ES mapping tools that have been reported in literature:

- ❖ Integrated Valuation of Ecosystem Services and Trade-offs (INVEST) **-** This is one of the most common ES mapping tools (Bagstad et al., 2013). It is freely available under the ownership of the Natural Capital Project. Its main aim is to model and map ES across the landscape to give a general picture of the patterns and changes in ES as influenced by land cover changes (Vigerstol and Aukema, 2011). INVEST consists of models used to map each ES. These ES models can also be used to explore the effects of future land use scenarios under different influencing factors. Its main input data is land cover data but also includes other data from look up tables or biophysical data. This tool produces quantitative ES maps expressed in either biophysical or monetary values.

A number of ES have been mapped at varying scales ranging from local to global level using this tool. Examples of ES that have been mapped using InVEST include: climate regulation (carbon storage and sequestration), reservoir hydropower production (water yield), Water purification (nutrient retention), avoided dredging and water purification (sediment retention), crop pollination, scenic quality (unobstructed views), recreation and tourism, managed timber production, wave energy production, offshore wind energy production.

- ❖ Artificial Intelligence for Ecosystem Services (ARIES) – This is also noted to be one of the most common ES mapping tools (Bagstad et al., 2013). ARIES is web based and allows users to evaluate trade-offs between ES and to identify remote ES beneficiaries. It relies on using Bayesian networks to show the relationships between input data and ES values. This tool produces a set of maps which show where ES are provided, the beneficiaries of those ES, the flow paths between source regions and use regions (Villa et al., 2009). It is applicable for mapping ES at a local to a global scale. The ES that have been mapped include: carbon sequestration and storage, aesthetic views, flood control, coastal flood regulation and open space proximity, freshwater supply, sediment regulation, subsistence fisheries and recreation.

- ❖ Land Utilisation and Capability Indicator (LUCI) **-** This is noted to be an extension of the Polyscape GIS framework used for analysing synergies and trade-offs among ES (Jackson et al., 2013b). It is designed to use simple algorithms. This tool categorises landscape elements into five service classes which range from very high existing value to very high opportunity for change. This tool mainly relies on DEM, slope, hydrography and land cover. The ES that have been mapped using this tool include: agricultural production, carbon, flooding, erosion, sediment regulation, water quality and habitat connectivity. This can be applied at a local to a catchment level scale.

- ❖ EcoServ-GIS – So far this tool is reported to be reliant on UK data sets (Bellamy and Winn, 2013). Its initial aim was to assist Wildlife Trusts decide on reserve management, inform policies and respond to local planning applications (Bellamy and Winn, 2013). It maps at a county scale the capacity of an ecosystem to supply an ES and also identifies areas of demand for the ES, and hence modelling the flow of ES. Capacity and demand models are created using look up tables or indicators of ecosystem processes. Available datasets such as the OS master map are used to create a base map assigning a habitat type to each parcel of land. The service models are then overlaid with the base map to identify ES flows, gaps and underutilised ES areas. The ES that have been mapped include: carbon storage, local climate regulation, noise regulation, pollination, water purification, ecological networks and a range of cultural ES e.g. accessible nature, aesthetics, community cohesion and education.

❖ Spatial Evidence for Natural Capital Evaluation (SENCE): This is a proprietary tool developed by a UK based Consultancy Company (Environment Systems Ltd). It uses an expert rule base system to assess, evaluate and map the effect of different habitat types in the delivery of ES (Vorstius and Spray, 2015, Medcalf et al., 2014). This tool utilises multiple datasets and uses the habitat map as the main input data. This tool has been applied at a local to a catchment level scale and the ES that have been mapped include: food, energy, timber, soil carbon, water quantity, water quality, vegetation carbon, erosion prevention, mitigation of flood risk, pollination, biodiversity, biodiversity resilience, landscape character, places of unique quality, history and archaeology, recreation and the ES within the marine environment.

In addition to the above described common tools, there are also many other ES tools that have been developed. These include:

➢ Social Values for ES (SoIVES) is a GIS based tool which is used to assess and map diverse stakeholder perceptions and values for ES (Sherrouse et al., 2011).
➢ Global unified metamodel of the biosphere (GUMBO) is aimed at modelling the dynamic linkages between social, economic and biophysical systems at a global scale (Boumans et al., 2002). Ecosystem services and how they contribute to human well-being are noted to be the main focus of this model.
➢ Multiscale integrated Earth Systems (MIMES) is a model used to evaluate the dynamics of ecosystem services given a number of future scenarios. Input data into the model is based on different spatial datasets and its output include spatially explicit time series of ES values. The models are used to determine the stock and flow of ecosystem services for particular cases.

Other scholars have evaluated these tools to assess their strengths and weaknesses. For example, Bagstad et al. (2013) reviewed 17 tools used to map ecosystem services and rated their performance against a set criteria to determine their strengths and weaknesses. Findings from their review showed that the tools greatly differed in their performance. Similarly, Vorstius and Spray (2015) evaluated SENCE, InVEST and EcoServ-GIS mapping tools to assess their potential as standard tools to map ES in local planning, using the Eddleston (approximately 70km^2) catchment as a case study. Their findings showed that these tools yielded varying map outputs influenced by factors such as the amount and type of data used to develop these. Vigerstol and Aukema (2011), compared tools for mapping freshwater ES and concluded that there is no best tool as they each had their own strengths and weaknesses, as summarised in the table below.

Table 2-3: Overview of advantages and disadvantages of different ES mapping tools

ES mapping tool	Scale at which ES are mapped		ES categories mapped				Aspects of ES mapped		Applicability		Open access	Software requirements		Capture of local variations in ES delivery
	Local	Multi-scale	Provisioning	Regulating	Cultural	Supporting	ES flows supply, demand areas	ES supply areas	Place specific i.e. UK use only	Any location including outside the UK		GIS	Web-based	
InVEST		√	√	√	√					√	√	√		
ARIES		√	√	√	√		√			√	√		√	
LUCI		√	√	√						√		√		
EcoServ GIS	√			√	√		√		√		√	√		√
SENCE		√	√	√	√	√		√		√		√		√
SolVES	√				√			√		√	√	√		√
GUMBO		√	√	√	√	√				√		√		
MIMES		√					√				√		√	

The ticks in the table above show the advantages of different ES mapping tools. As shown in the table above, some ES mapping tools such as SENCE can be used to map ES supply areas at multi-scales, it can be used to map the four main ES categories and adjusted to capture local variations in ES supply. However, as a proprietary tool, SENCE has a disadvantage of not being freely available. Other ES mapping tools on the hand, are mostly open source and offer users the advantage of unrestricted access.

Clearly, the preceding sections show that there is neither a best method nor a best tool to map ES, but each has its own strengths and limitations. The choice of the mapping method used can vary from one place to the other and is determined by the purpose of ES mapping though other commentators (Vigerstol and Aukema, 2011, Maes et al., 2013, Pagella and Sinclair, undated) note that in most cases it is largely influenced by data availability and intended scale of mapping. Similarly, tools developed map ES at varying scales using varied multiple datasets, some of which are applicable to any location while others are place/site specific (Bagstad et al., 2013).

Seppelt et al. (2011) consider the varied nature of these approaches as an indication of the fragmented nature of the ES research area and arguably a weakness thereof. Consequently, many commentators (Maes et al., 2013, Maes et al., 2012a, Martínez-Harms and Balvanera, 2012, Crossman et al., 2013) call for a standardised and consistent methodological approach to mapping ES. These authors are of the view that such a standardized approach would ensure consistency in ES mapping approaches and allow for comparison of ES mapping results across countries and regions. For example, in the EU, the MAES working group was set up within the Common Implementation Framework of the Biodiversity 2020 Strategy as a way of standardizing ES mapping approaches within member states (Maes et al., 2013). This working group proposed a conceptual framework for EU wide ecosystem assessments (see Figure 2-6, page 27) and mapping based on CORINE land cover classes and the Common International Classification of Ecosystem Services. Crossman et al. (2013) further propose the need for a standard process for documenting ES mapping and modelling studies (a blueprint).

Grêt-Regamey et al. (2015), however, caution that calling for standardized mapping approaches needs to be considered carefully as there is no "*cure-all*" approach that can address the many varied and complex interactions and factors associated with socio-ecological systems. These authors instead propose a four step flexible and adaptable approach to mapping ES which involves: firstly, defining the goal of the ecosystem service mapping, followed by a meta-analysis of relevant ES mapping studies to identify key variables for mapping the selected ecosystem services. Thirdly, the identified variables can be attributed to different levels of the multitier framework according to the level at which they best answer the posed research or policy questions. Lastly, appropriate methods for mapping the ecosystem services can be selected based on reviewed studies. Such suggestions, however, remain debatable.

2.4.4 Which ecosystem service types are most frequently mapped?

Both the number and type of ecosystem services mapped vary between studies. The review by Martínez-Harms and Balvanera (2012) showed that 19 different ES have been mapped by different studies, while Crossman et al. (2013) observed an average of 5.6 ES mapped per study. Seppelt et al. (2011) found that nearly 50% of the studies reviewed mapped, at a time, only five or fewer ES with 19 being the highest number of ES mapped.

Regulating services such as carbon storage are the most frequently mapped ES (Martínez-Harms and Balvanera, 2012, Egoh et al., 2008, Crossman et al., 2013, Malinga et al., 2015). These are followed by provisioning, cultural and supporting services respectively (Malinga et al., 2015, Crossman et al., 2013). This has resulted in a bias towards mapping regulating and provisioning ecosystem services especially those that are strongly influenced by land cover and less coverage of cultural ones.

Reasons for more studies mapping regulating and provisioning ecosystem services have been mainly attributed to their predominance and importance to stakeholders in an area. For example, Kandziora et al. (2013) mapped agricultural related provisioning services, i.e. crops and fodder for livestock production, as these were reported to be the dominant activity in their study area in Northern Germany. Similarly, the ES mapped in the Scottish Borders under the LUS pilot project were those that were considered to be important from stakeholder consultations and prioritised ES in the Scottish Borders (Spray, 2014). The ES mapped by Egoh et al. (2008) in South Africa were considered to be of national relevance, while the ES mapped by Crouzat et al. (2015) were those ecologically, socially and economically relevant to the French Alps. Petter et al. (2013) also mapped important ecosystem services in South East Queensland (Australia), while Vihervaara et al. (2010) mapped ecosystem services that were of concern and importance to the community in their study area in Finland.

The choice of ecosystem services mapped is also influenced by data availability. For example, Leh et al. (2013) selected and mapped those ecosystem services for which they could access data. Regulating services, especially carbon storage are also frequently mapped, arguably because of increased awareness and recognition of their importance in climate regulation in face of ongoing environmental challenges associated with climate change. On the other hand, mapping cultural ecosystem services is noted to be challenging compared to these other ecosystem service categories, as some cultural ecosystem services involve individual subjective judgements (Hauck et al., 2013b).

2.4.5 Scales for mapping ecosystem services

Martínez-Harms and Balvanera (2012), found that most ES were mapped at the regional and national scale and less at local and global scales. A recent review by Malinga et al. (2015) showed that the majority of studies (53%) were done at the municipality scale, which denotes an increase in mapping of ecosystem services at lower scales. This could be understood as a move towards mapping ecosystem services at scales that are considered important for decision making and where policies are implemented.

There are calls for more studies to map ecosystem services at local scales (Kandziora et al., 2013) to inform decision making. Local scales in the case of freshwater ecosystems equate to a catchment level scales. As discussed earlier, catchments are acknowledged to be appropriate units for the management of freshwater ecosystems as they allow for the integration of land use and water resources management. This means that mapping ecosystem services at this scale would also assist in the implementation of directives, such as the WFD. This could be through for example the involvement of relevant stakeholders who could further assist towards the identification and prioritisation of ecosystem services supplied by river catchments. Despite the importance of local scales in mapping ecosystem services, in current practice, data availability influences the scale at which ecosystem services are mapped (Malinga et al., 2015).

2.4.6 Assessment of changes in ecosystem services over time

Singh (1989), defines change detection as the process of identifying differences in the state of an object or phenomenon by observing it at two or more time points. This could be to determine land cover/ land use change, habitat change, landscape change, urban changes, environmental changes, etc. As noted by Lu et al. (2004) understanding change is important in: (1) understanding interactions between human activities and the natural environment, (2) monitoring and managing natural resources and (3) promoting better decision making. Similarly, understanding changes in ecosystem services over time can deepen understanding on how they can be managed.

Mapping ecosystem services is at the core of current contemporary approaches to assessing changes in ecosystem services. As discussed in the previous section, this is partly due to the use of habitat/land cover proxy data for mapping ecosystem services. As a result assessment of changes in ecosystem services over time is biased towards change detection techniques used in habitat/land cover mapping as for example, was done by Burkhard et al. (2012), Lautenbach et al. (2011) and Metzger et al. (2006). Analysis of changes in ecosystem services also depend on metrics and indicators used to map these and this differs among studies (Nemec and Raudsepp-Hearne, 2013).

Among these, GIS is noted to provide a powerful tool in the visualisation and analysis of ecosystem service delivery in landscapes (Baral et al., 2009). In GIS, approaches to estimating changes in ecosystem services include the generation of static estimates presenting a snapshot of current or past ecosystem services across a landscape (Nemec and Raudsepp-Hearne, 2013). GIS is used as a tool both to visualise and to quantify changes in ecosystem services supply areas (Andrew et al., 2015).

2.5 Section 4: Chapter summary

A broad literature review has revealed increasing interest in both policy and science towards the ecosystem services concept for the management of the natural environment, including threatened and degraded ecosystems such as river catchments. This is however, not to say this concept is intended to supersede existing conservation and environmental management approaches. Rather, it is perceived as adding value to these, especially through illustrating the link between nature and human well-being and explaining this in ways that can be easily understood. However, as discussed earlier such a supposition continues to be a subject of huge debate.

There are clearly a number of differing views and inconsistencies within the ecosystem services research area. Such inconsistencies range from the definition and classification of ecosystem services, through to mapping approaches and tools used, including the number and type of ecosystem services mapped, as well as the scales at which these are mapped. This can be interpreted as revealing the fragmented nature and weakness thereof in the ecosystem service research area. On the other hand, it can be understood as denoting progressive development and improvement in this research area, reflecting the flexible and adaptable nature of this concept.

Optimism on the potential utility of the ecosystem services concept in managing the natural environment is evident. However, debate and deliberations remain. Some, including Norgaard (2010) view it as too simplistic to inform policy making on complex environmental processes yet others such as Plant and Ryan (2013) consider it as offering a simple way of engaging with stakeholders and policy makers.

Mapping of ecosystem services was highlighted as an emerging research area of the ecosystem service concept. It is considered a vital aspect of this concept and currently at the core of contemporary approaches to assessing changes in ecosystem services. To do this, GIS has been increasingly used to visualise ecosystem service supply areas in landscapes such as catchments, as well as used to analyse changes in ecosystem service supply over time. The next chapter provides details of the research methodology followed to collect and analyse data, including the justification for the selection of such methods.

3 Methodology

3.1 Chapter Introduction

This chapter presents the research methodology followed in this study. It is divided into four main sections. The first section briefly describes the conceptual framework informing this study and the study areas chosen, including justification for their selection. It also gives an outline of the approaches used to detect change in habitats and ecosystem services. An explanation and justification for selection of historic aerial photographs as the main data source used in this study is also included.

The second section provides a detailed description of the data collection and processing stages, starting from searching and acquiring the historic air photos to how they were used to reconstruct the historic landscapes for the study areas and how they were interpreted to derive the historic habitat maps through to mapping historic ecosystem services. Reasons for the preference and adoption of selected methods in each of the three stages are also provided.

The third section describes the sources of uncertainty, error and data limitations associated with each of the data collection stages. Also included are measures implemented to assess the accuracy with which data processing was done and maps generated. It is summarised with an error budget illustrating the inherent and collective sources of uncertainty accrued during the entire data collection and processing process.

The final section highlights the key outcomes and how these inform subsequent chapters.

3.1.1 Methodology

Figure 3-1 below provides an outline of the overall methodological approach to this study. It illustrates how the identified knowledge gaps, presented in the previous chapters informed the formulation of research questions that this study sought to address. These in turn informed the data collection and processing methods adopted and subsequent data analysis and interpretation approaches used.

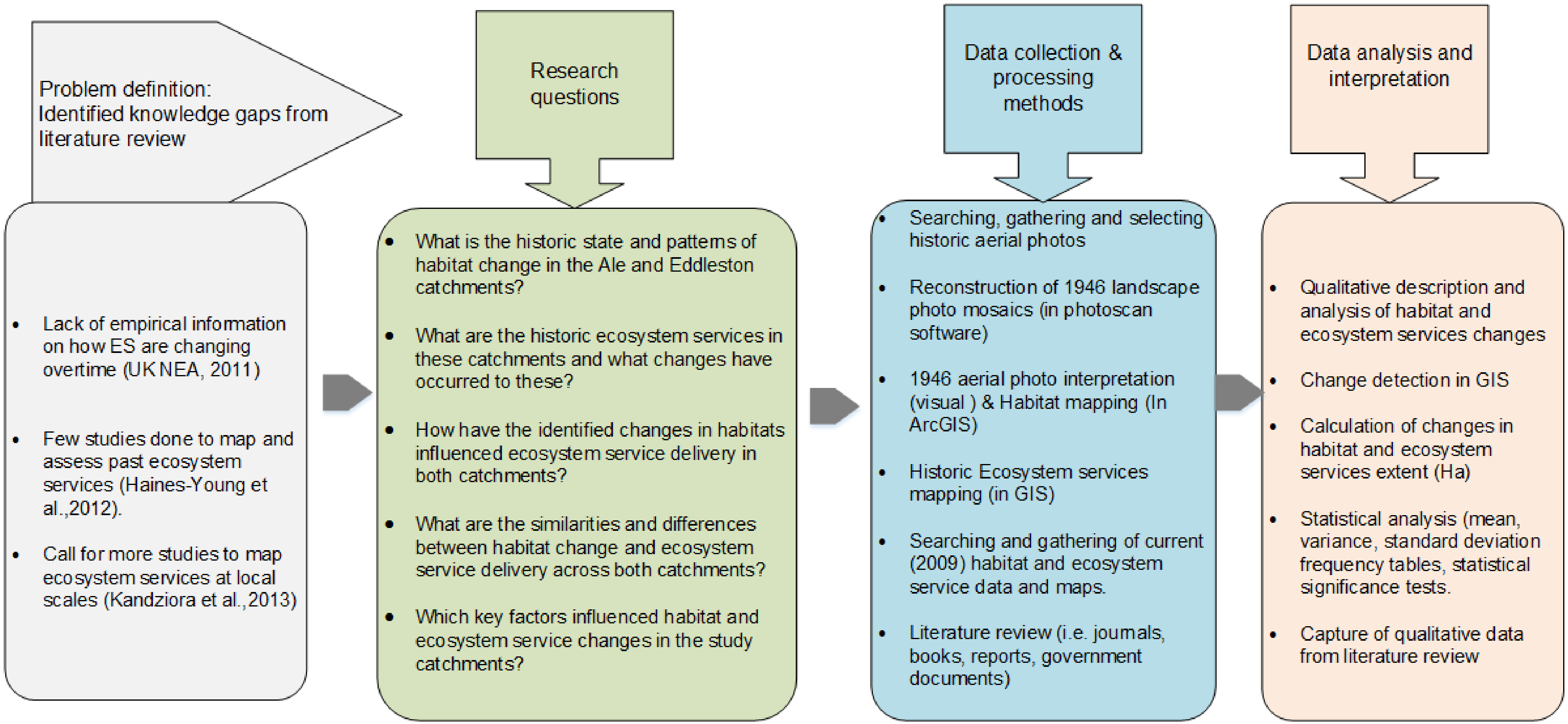


Figure 3-1: Study Methodology

3.1.2 Conceptual framework: The Ecosystem service cascade

As discussed earlier, the "ecosystem services cascade" has been used as a conceptual framework to understand ecosystem services (Haines-Young and Potschin, 2010) and has since been used as a common conceptual framework for assessing and mapping ecosystem services, as it provides a stepwise illustration of the connection between ecosystems and human well-being (Maes et al., 2012a). On this basis, the ecosystem services cascade was considered a suitable and appropriate framework through which the research aims of this study could be addressed and was modified (Figure 3-2) in line with the intentions of this study. As shown in figure 3-2 below, ecosystem services link ecosystems and socio-economic systems. Ecosystems are here represented by broad habitat types, based on the assumption used in many current proxy based ecosystem services mapping practices which equate these to different distinct ecosystems.

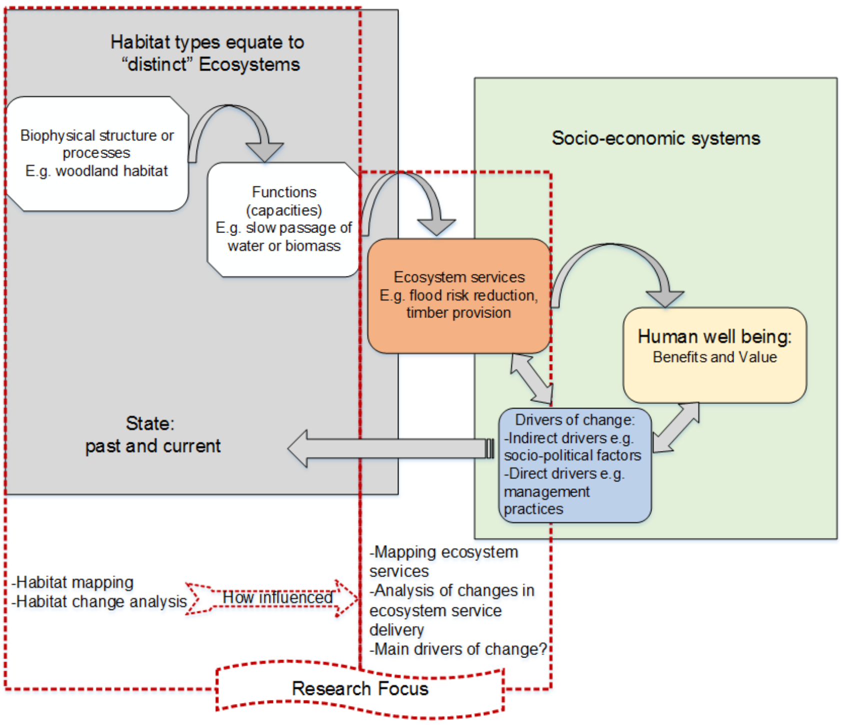


Figure 3-2: Ecosystem services cascade model as a conceptual framework to map and assess spatial changes in ecosystem services
(*Source: Modified from Haines-Young and Potschin (2010)*)

As illustrated in the figure above, ecosystem processes and functions influence the delivery of different ecosystem services which are of benefit to different components of human well-being. Figure 3-2 also shows a feedback loop showing the impact of human actions on both the past and current state of ecosystems through direct or indirect drivers of change. Such drivers of change impact on ecosystem processes and functions which in turn impact on ecosystem service delivery. The influence of these factors either enhances or degrades ecosystem functions, in turn impacting on ecosystem service delivery. These hypothesized relationships provided a basis for mapping ecosystem services and assessing changes in ES delivery over time.

3.1.3 Study Area

The study areas chosen were two sub catchments (Ale and Eddleston) of the Tweed catchment (Figure 3-4). The Tweed catchment (Figure 3-3) covers an area of about 5000 km^2 and forms part of the border between Scotland and England (Collins, 2004). It is located to the south east of Scotland and north east of England, with about 16% of this catchment lying in England. The catchment extends from the uplands of the Lammermuir Hills in the north, the Southern Uplands in the west and the Cheviots Hills in the south through the valleys of the Tweed, Teviot and Till, to the town of Berwick-Upon-Tweed in the east. The Tweed River flows 160km before discharging into the North Sea. The area of the Tweed catchment in Scotland is also referred to as the Scottish Borders and administratively falls under the Scottish Borders Council.

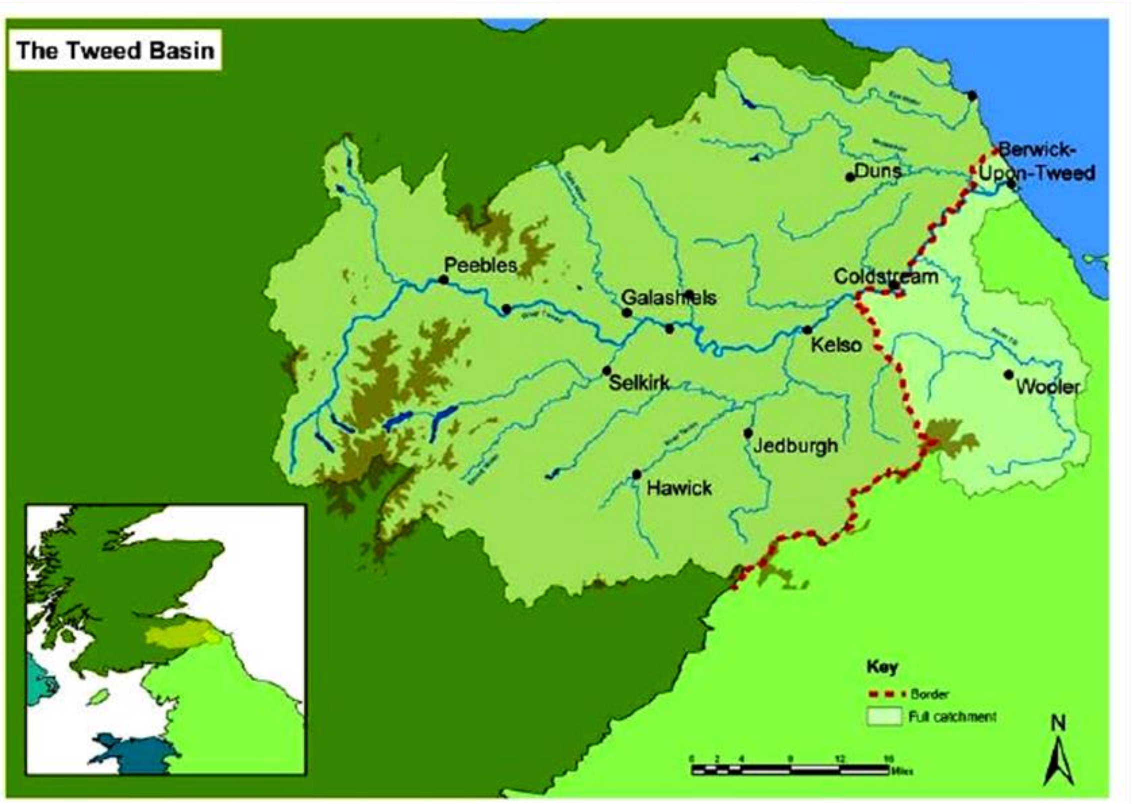


Figure 3-3: The Tweed catchment
Source: www.tweedforum.org

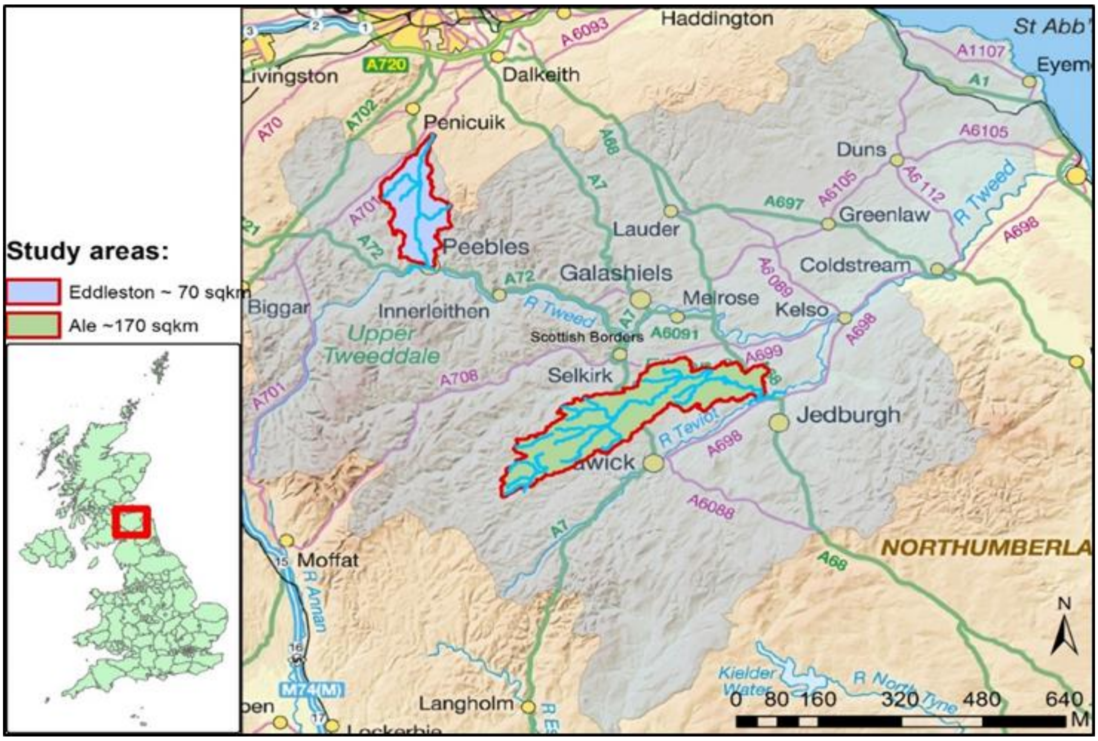


Figure 3-4: Location of the Ale and Eddleston sub catchments within the Tweed catchment
Contains OS data © Crown Copyright 2015

The Ale and Eddleston sub catchments (Figure 3-4) of the Tweed were selected for this study because:

1) they are within the same catchment and share a similar long history of past land use and water management changes due to drainage, infilling and improvement of catchments for agricultural purposes, dating back to the 18th century (Harrison, 2012). It was of interest to understand the extent of impact of these past land use and water management practices on catchment landscapes;

2) dominant land uses in these sub catchments have resulted in different issues and pressures, the key ones as identified by SEPA include: (1) nutrient enrichment from diffuse pollution resulting from farming activities, forestry and land development all contributing to pollution of water bodies in these catchments, (2) fragmentation of habitats such as wetlands, (3) abstraction of water for public water supply, and (4) increased flood risk related to alteration of river beds, bank (hydro-morphological alterations) e.g. through channelization, upstream land use, past water management approaches and floodplain developments;

3) data allowing for a historic assessment of change in habitats and ecosystem services was available. Of importance was the availability of current habitat and ecosystem services maps, verified by local stakeholders. Both catchments were part of the six sub catchments selected for stakeholder engagement and ES mapping during the pilot Land

Use Strategy project (Spray, 2014). Such maps were accessible and available to compare and assess change in habitats and ES over time; and

4) there is previous and ongoing research work in both catchments which the historic focus of this study could augment and build on.

3.1.3.1 The Ale sub catchment

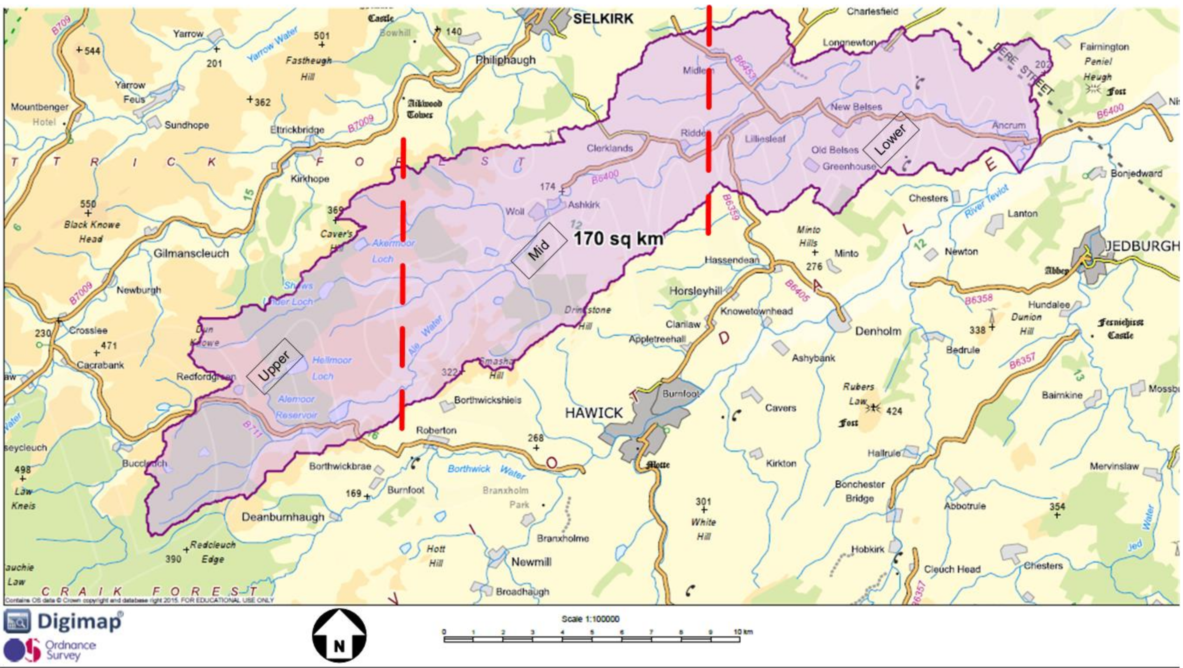


Figure 3-5: The Ale catchment

The Ale catchment covers an area of about 170 km^2, and can be arbitrarily divided into three sections, the upper, mid and lower catchment areas (figure 3-5). The lower section of this catchment are its low lying areas, dominated by agriculture farmland and settlement areas such as Ashkirk and Ancrum. In line with River Basin Management Planning under the WFD, the Scottish Environment Protection Agency (2010) characterized the overall condition of surface waters in the lower section of the Ale catchment as moderate. The physical condition of the surface waters was classified as good, while the water quality was classified as moderate (Scottish Environment Protection Agency, 2010). Main pressures in this part of this catchment include nutrient enrichment and diffuse pollution from agricultural activities and rural land use.

The middle section of the Ale catchment consists of both undulating hilly areas and low lying areas (marginal areas). It is mainly dominated by reservoirs and semi-natural habitat types associated with the uplands. The overall condition of surface waters in this section of the Ale was classified as good. The physical condition of the Ale water was also characterised as good with high water quality (Scottish Environment Protection Agency, 2010).

The upper section of the Ale catchment are its uplands areas, dominated by forestry plantations, reservoirs as well as semi-natural habitat patches of heath, bracken, acid grassland and different wetland habitats e.g. bogs. The overall condition of surface waters in this part of the catchment was characterised a poor due to abstraction activities for public water supply and forestry activities. However, the physical condition of the surface waters was classified as good with high water quality (Scottish Environment Protection Agency, 2010).

According to Tweed Forum (2013), the main issue in the Ale catchment is related to fragmentation of wetland habitats. This impacts on their connectivity and wetland biodiversity, making the wetlands vulnerable to changes in their surroundings and reducing chances of species exchange and colonisation between habitats (Medcalf and Williams, 2010). Following the recent recognition of the conservation and recreational value of wetlands in this catchment, there is increased interest among local farmers on how the management of such wetlands can contribute to farm diversification and income generation (Tweed Forum, 2013).

3.1.3.2 Eddleston sub catchment

The Eddleston catchment covers an area of about 70 km^2. The catchment is bordered by Moorfoot Hills to the east and the Cloich Hills to the west, the uplands areas of this catchment. The Eddleston River is bordered by these uplands on both sides and flows within the low lying areas of this catchment. It joins the main stem of the Tweed River in Peebles (Figure 3-6).

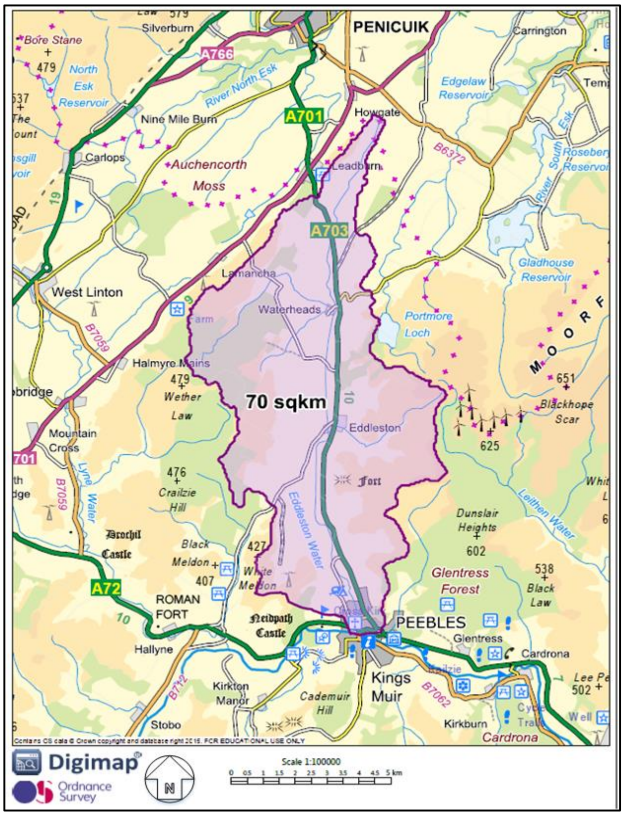


Figure 3-6: The Eddleston catchment

Harrison (2012) outlines the historic changes to this catchment and notes that much of the main stem of the Eddleston Water was straightened some 200 years ago between Peebles and Edinburgh. Due to these historic changes and upstream land use, the main issue in this catchment is increased downstream flooding, especially at Peebles (Tweed Forum, 2015a). The Eddleston catchment was initially categorised by SEPA as “poor” under the WFD river basin management planning due to historic hydro-morphological alterations and channelization of the Eddleston water (Werritty et al., 2010). However, the ecological status of this river was recently upgraded from “poor” to “moderate” under the WFD characterisation (Tweed Forum, 2015a). This is due to a number of Natural Flood Management (NFM) initiatives being implemented to restore the physically degraded reaches of the Eddleston Water through for example, re-meandering sections of this river and introduction of riparian woodland plantations in the upstream catchment areas. This catchment has been the focus of much research on natural flood management and river habitat restoration opportunities (Werritty et al., 2010, Spray et al., 2010).

3.1.3.3 The Tweed catchment

The Tweed catchment itself is the focus of several initiatives on land use and catchment management and this provides a suitable context for this study. The Tweed catchment has diverse uses, habitat types and designations reflecting its importance in multiple ecosystem service provision, including supporting livelihoods and contributing to the local economy (Tweed Forum, 2010).

Typical to rural Scotland, 75% of the Tweed catchment is under agriculture, alongside other land uses such as forestry. Tourism and fisheries are also other major activities in the catchment. Current main uses of water in the Tweed catchment include water supply for the Scottish Borders and Edinburgh and agricultural irrigation. Other recreational activities done in this catchment include walking, bird watching, fishing, sailing, swimming, camping, educational visits, mountain hikers among others. Human population, mainly settled in valley towns is estimated to be about 130 000 (Spray and Comins, 2011).

This catchment is subjected to a number of pressures related to its diverse and competing uses which, as identified by SEPA main ones include: (1) increased use of pesticides, nutrient enrichment from diffuse pollution resulting from farming activities, forestry, land development and industry all contributing to pollution of water bodies in the lower part of this catchment, (2) increased flood risk related to alteration of river beds, bank (hydro-morphological alterations) e.g. through channelization, upstream land use, past water management approaches and floodplain developments, (3) fragmentation of habitats such as wetlands, (4) abstraction of water for public water supply and farming and (5) increased presence and risks associated with invasive non-native species such as the American Signal Crayfish which poses a threat to the salmon fisheries.

In line with River Basin Management Planning under the WFD, the Scottish Environment Protection Agency characterized the condition of surface waters in this catchment as: having just over half (52%) of these in a good or better condition, 38% of these are moderate and 10% poor, some waters were classified as heavily modified through engineered flood regulation, channelization (Scottish Environment Protection Agency, 2010).

Tweed catchment: with a well-established stakeholder led NGO – the Tweed Forum

The Tweed catchment has a well-established stakeholder led NGO - the Tweed Forum. The Tweed Forum which began in 1991 has been largely known as a successful and active stakeholder led organisation which has been instrumental in the management of this catchment (Tweed Forum, 2015b). Spray and Cumins (2011), outline the origin of this NGO and reflect how it evolved as a local community led initiative before the statutory requirements such as AAG under the WFD RBMP and has emerged as one of the leading stakeholder led NGOs involved in catchment management and stakeholder engagement. Its main role has been to empower local communities to achieve their ambitions and it has over the years gained local trust from stakeholder communities. It has also played an essential role as the intercessory body between local stakeholder communities, the policy makers and authorities in the catchment (Cook et al., 2013). For example, Tweed Forum works with SEPA to facilitate the implementation of measures aimed at restoring good ecological status of water bodies in this catchment. It has also; working with the Scottish Borders Council, University of Dundee etc., been instrumental in facilitating local stakeholder involvement in the pilot Land Use Strategy as well as the on-going natural flood management measures in the Eddleston catchment (Tweed Forum, 2015b). The existence of such a platform can be useful in this research especially in providing the evidence base that could inform its stakeholder engagement work, policy implementation and linking with the principles of the HELP programme.

Other than supporting implementation of policy intentions, Tweed Forum also works in implementing the Tweed Catchment Management Plan, which was put in place together with responsible organisations to ensure the equitable and sustainable utilisation, conservation and protection of the water resources in the catchment (Tweed Forum, 2010).

Tweed catchment as a UNESCO-HELP basin

The Tweed catchment is a UNESCO- HELP basin, providing a base for science evidence based decision making, catchment management and policy implementation. The Tweed catchment was designated a UNESCO-HELP basin in 2008 (Hendry, 2008). HELP is the UNESCO`s Hydrology for the Environment, Life and Policy programme whose aim is to improve the links between hydrology and the needs of society (UNESCO, 2015). In so doing, this programme is aimed at conducting research on water resources management that would deliver practical benefits for stakeholders. This promotes integrated catchment management underpinned by science while also establishing the link between policy and practice. In this case science is expected to provide the evidence base needed to address issues identified by stakeholders while also informing policy implementation. Such an understanding between science, society and policy would inform implementation of water management actions based on jointly agreed solutions, more so in face of increasing environmental challenges in this catchment. This study would form an important contribution towards this cause of the HELP programme while also presenting this catchment as an exemplar study area where sound science evidence base can inform catchment management.

3.1.4 Change detection in habitat and ecosystem service delivery based on air photo interpretation

Key aspects of change detection applicable to habitat/land cover or landscape change analysis have been identified by Lu et al. (2004) and Macleod and Congalton (1998). These include (1) detecting a change has occurred, (2) identifying the nature of change, (3) measuring the areal extent of change, and (4) assessing the spatial pattern of change. This section focusses on the first aspect i.e. detecting a change has occurred while the next chapter (results) presents the other three aspects of change detection.

Figure 3-7 outlines the habitat and ecosystem services change detection approaches undertaken in this study. Change detection was based on two dates and two change detection methods were combined to detect changes in habitat and provide a basis for assessing changes in ecosystem service delivery. The two approaches used were: (1) onscreen digitizing and visual interpretation of historic aerial photography and (2) use of GIS and other ancillary data sets to aid air photo interpretation through overlaying and analysing these in GIS. GIS was also used to assess changes in ecosystem service delivery. Reasons for selection of these approaches are explained in detail in respective sections of this chapter but were basically informed by the historic focus of this study and their applicability.

Visual analysis was for example, preferred over automated classification approaches as this study relied on black and white aerial photography with one band of gray and limited spectral variations for habitat mapping which otherwise could have been a challenge if automated classification approaches were used. Lu et al. (2004) notes that the use of automated techniques such as supervised clasification for historic studies is a challenge as temporal equivalent training data required for these techniques is a hardly available.

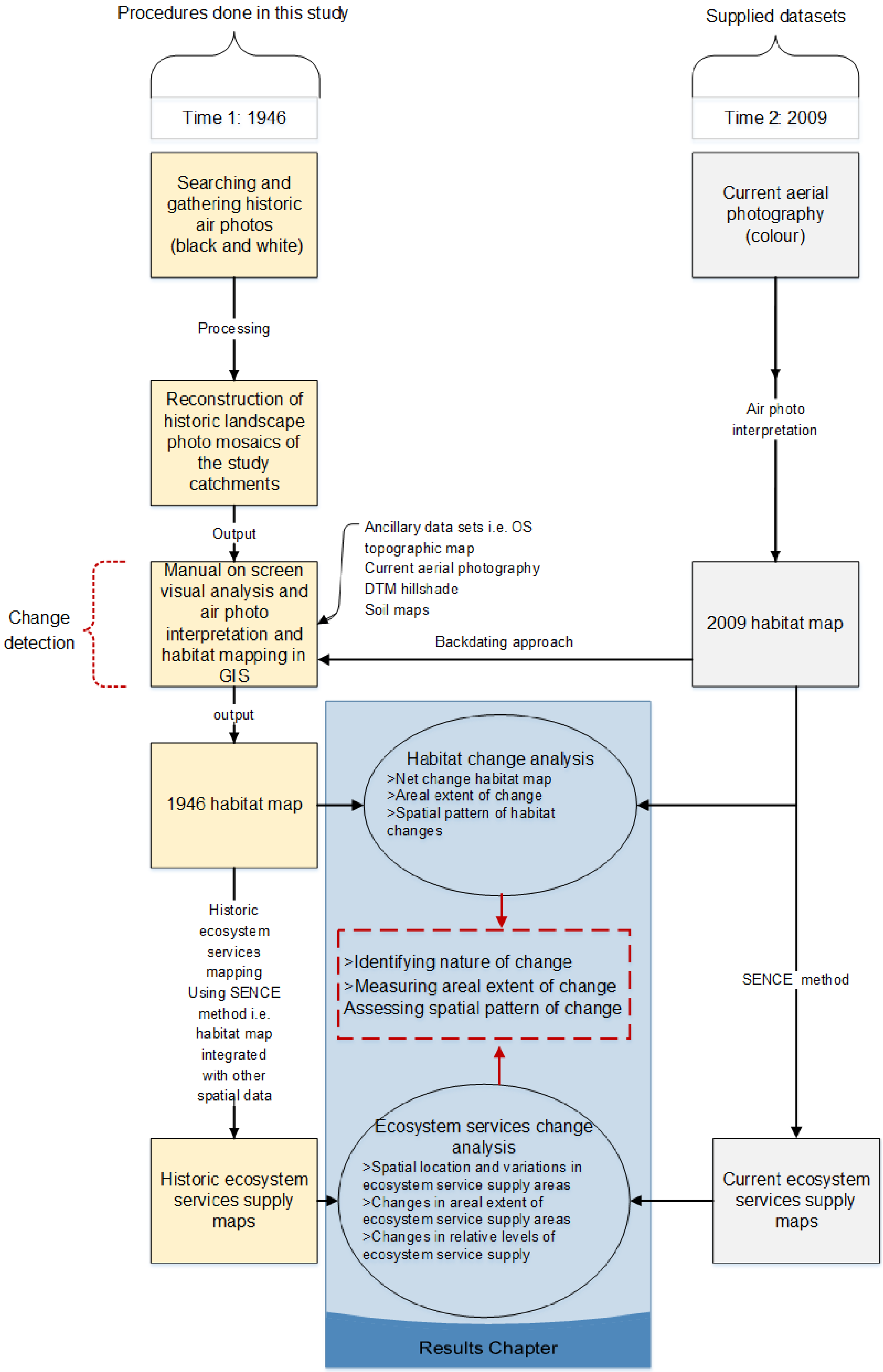


Figure 3-7: Change detection and analysis of habitat and ecosystem services procedures undertaken in this study

As illustrated in the figure above, the focus of this study was on collecting, processing and interpreting data for historic habitat and ecosystem services mapping in the study catchments. Current data was supplied under agreement from the Scottish Borders Council. The current habitat maps covering the study catchments, were produced through aerial photography interpretation by Environment Systems Ltd as part of network and opportunities mapping for habitat restoration under the Scottish Borders` Local Biodiversity Action Plan. The current ecosystem service maps were produced as part of the Land Use Strategy pilot project in the Scottish Borders, a part of which Environment Systems Ltd was contracted to do the baseline ES mapping from aerial photography and Phase 1 habitat classification system.

3.1.5 Main data source: Aerial photography

This research utilises aerial photographs to reconstruct historic landscape photo mosaics of the Ale and Eddleston catchments in order to assess the type and extent of habitat changes and subsequently how these changes translate to changes in ecosystem service delivery in these areas. In order to detect these changes at a local catchment scale, aerial photography acquired at different time periods was considered to be an appropriate data source. This is because, the high spatial resolution of this dataset allows for detailed mapping of habitats (Cherrill and McClean, 1995) or, more broadly the assessing and mapping of landscape change over long time scales (Taylor et al., 2000, Csaplovics, 1992, Casson et al., 2003, Kull, 2005, Turner and Ruscher, 1988, Morgan et al., 2010) In addition, air photos are not limited by atmospheric distortion, compared to satellite imagery (Juel et al., 2013).

While the use of satellite derived land cover data like CORINE is widely accepted in current ecosystem services mapping practice, Verhagen et al. (2015) noted that the spatial resolution of this data set might not detect small sized but substantial habitat types such as hedgerows, which are important in ecosystem service delivery. By comparison, the high resolution of air photos can allow for detection of such features. Furthermore, aerial photographs are among the most common traditional data sources for historic assessments (Lu et al., 2004, Cots-Folch et al., 2007) and have been available for a long time period, well before the advent of satellite imagery (Li and Shao, 2011, Kull, 2005).

A number of studies have utilised aerial photography to assess habitat/LC/LU change over time. Examples of such studies include the one by Thomson et al. (2007) in which land cover change between 1950,1990 and 2000 around Natura 2000 sites in the UK was assessed through the use of historic and recent aerial photography. Taylor et al. (2000) interpreted historic aerial photographs to assess landscape change in the National Parks in England and Wales. Jauhiainen et al. (2007) used aerial photographs from 1995 and 1946 to monitor changes in peat land ecosystems in Finland. A study by Halpern and Meadows (2013) utilised aerial photographs from 1960, 1977, 1988, 2001 and 2010 to quantify land use changes in Swartland, Western Cape in South Africa. Newton and Knight (2005) used a series of aerial photographs taken at ten year intervals since 1938 to assess agricultural changes of the west coast renosterveld in South Africa. The Macaulay Land Use Research Institute (1993) (Now James Hutton Institute) compiled the land cover of Scotland 1988 (LCS88) dataset through aerial photography interpretation. In Denmark, aerial photographs were interpreted to collect forest information which was used in modelling the Danish landscape (Groom et al., 2006). Gerard et al. (2010) also used aerial photography to assess land cover change in Europe between 1950 and 2000.

Aerial photographs have also been used in combination with other data sources such as topographic maps and satellite imagery to assess ecological changes. For example, Chadwick et al. (2005) utilised aerial photographs and satellite imagery to study the historical motion of the Salmon Falls landslide in Idaho, USA. Csaplovics (1992) used aerial photography and SPOT satellite data to monitor land cover change of a heathland region in France. A study by Anna (2003) was based on the visual interpretation of IKONOS 2000 satellite imagery and 1975 aerial photographs to detect vegetation changes on the Swedish Mountainous Heaths. Eremiasova and Skokanova (2009) used old topographical maps and aerial photographs to map land use changes in the Czech Republic. Cousins (2001) compared aerial photographs with 17th and 18th century cadastral maps to analyse land cover change in Sweden. Johansson et al. (2008) used old maps and aerial photographs to investigate changes in the distribution and extent of semi-natural grasslands in the Oland landscape in Sweden.

Set against the above, aerial photographs were considered appropriate for reconstructing the historic ecological landscape of the selected catchments as they can provide the desired ecological detail. The following sections detail how aerial photographs were collected, processed and analysed inline with the aims of this study.

3.2 Data collection and processing methods

Data collection and processing was divided into three linked stages. The first stage was on searching, gathering and selection of historic air photos covering the Eddleston and Ale catchments. These were subsequently aligned to reconstruct the landscape photo mosaics (orthophotos) of these catchments. The photo mosaics were then visually interpreted in ArcGIS (stage 2) to derive the 1946 habitat maps for these study areas. The final stage (stage 3) included mapping historic ecosystem services in these study catchments. This was based on a rule based approach that translated the 1946 habitat maps into indicative ecosystem service maps. The figure below is a flow chart illustrating the data collection process.

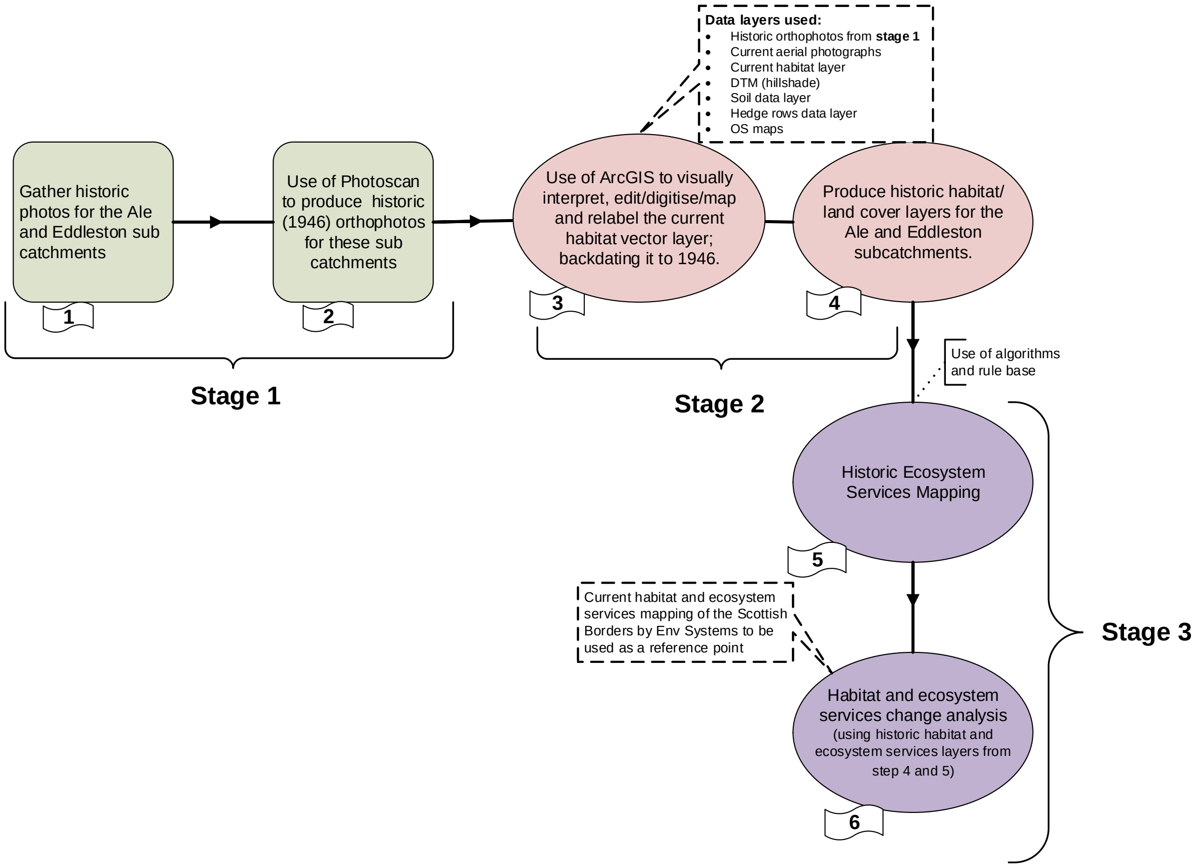


Figure 3-8: Data collection and processing flow chart

As shown in the flow chart above, output from one stage was an input in the next consecutive steps. In total, the data collection, synthesis and manipulation process took 18 months. This process took much longer than anticipated due to: (1) the time required to locate and access necessary historic air photos, (2) the time required to obtain relevant licences, and (3) the subsequent need to access specialist computing facilities which were required to handle the amount of data collected and allow for its processing and manipulation. In addition, data processing was an iterative process which at times required a repetition of the previous stage to improve the accuracy of the outputs and to ensure that the next stage is correctly done.

3.3 Stage 1: Gathering and processing of historic air photos

3.3.1 Searching, gathering and selection of historic air photos for the study areas

The main criteria for choice in searching for historic air photos was to get photos that covered an important period influencing habitat and ecosystem service changes in the study catchments. Given that ecosystems take long to respond to change, a time period giving such a long time span was considered and also with good coverage of the study catchments. These also had to be of good quality and suitable for analysis in the subsequent processing stages.

Ale catchment

After exploring the availability and possibilities of accessing the historic air photos for the selected study catchments, a collection of black and white aerial photographs starting from the 1940s for the whole of the Scottish Borders was acquired under agreement, from the Tweed Forum and Scottish Borders Council. These air photos were extracted from the Royal Air Force (RAF) surveys from the 1940s done for training purposes and probably to also inform planning and reconstruction work following the devastations of the Second World War. The air photos were archived in 22 CDs. The acquisition dates and number of photos in each of the CDs varied, with some having as much as 1500 photos while others had 300 or less, not in any sort of order. This collection of photos was complemented with flight lines and sortie plots data (GIS format) which showed the aerial view of the flight paths and the points where the photos were captured during the surveys. However, taking of these air photos during the surveys was unsystematic as illustrated in the flight paths figure below.

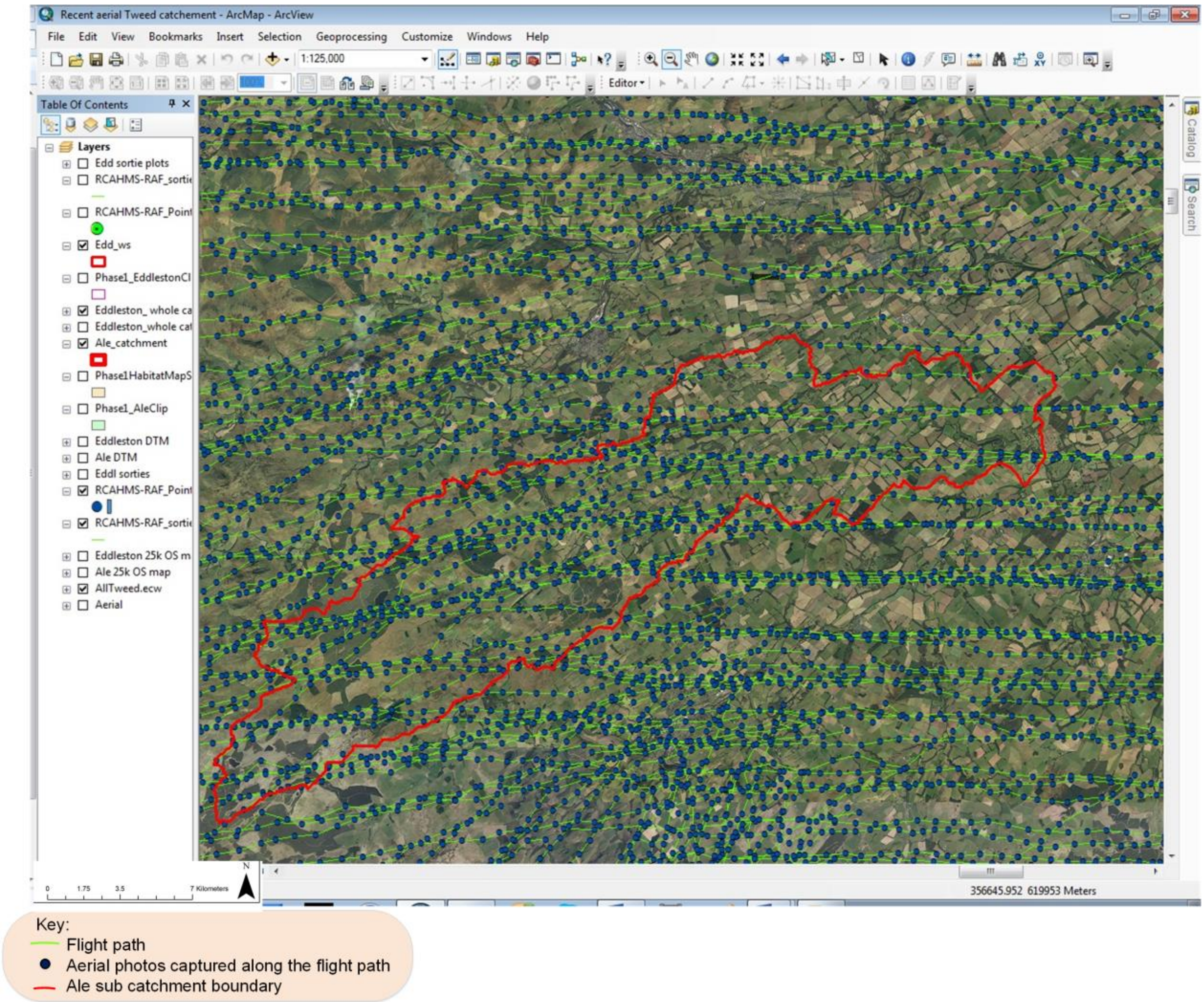


Figure 3-9: Unsystematic capture of the air photos during the Royal Air Force surveys

The lines in figure 3-9 show the flight paths during different time periods of air photo capture and the dots represent photo capture positions. Not all areas of the catchment were completely covered during these surveys. Some of the areas were captured during the surveys done much later in the 1950s and 1960.

Air photos were selected that covered the period (1940s) and area of interest i.e. the Ale sub catchment (figure 3-10 shows how the selection process was done). The selection of photos was done with the aid of Ordnance Survey (OS) maps and recent colour aerial photography which were used for matching and selecting corresponding black and white photos covering the area of interest. Assistance in verifying the selected photos was also provided by an experienced Project Officer from Tweed Forum who is familiar with the catchment. In total 385 digitally scanned unregistered photos (scale 1: 10 000) were selected and used to reconstruct the landscape photo mosaic of this sub catchment.

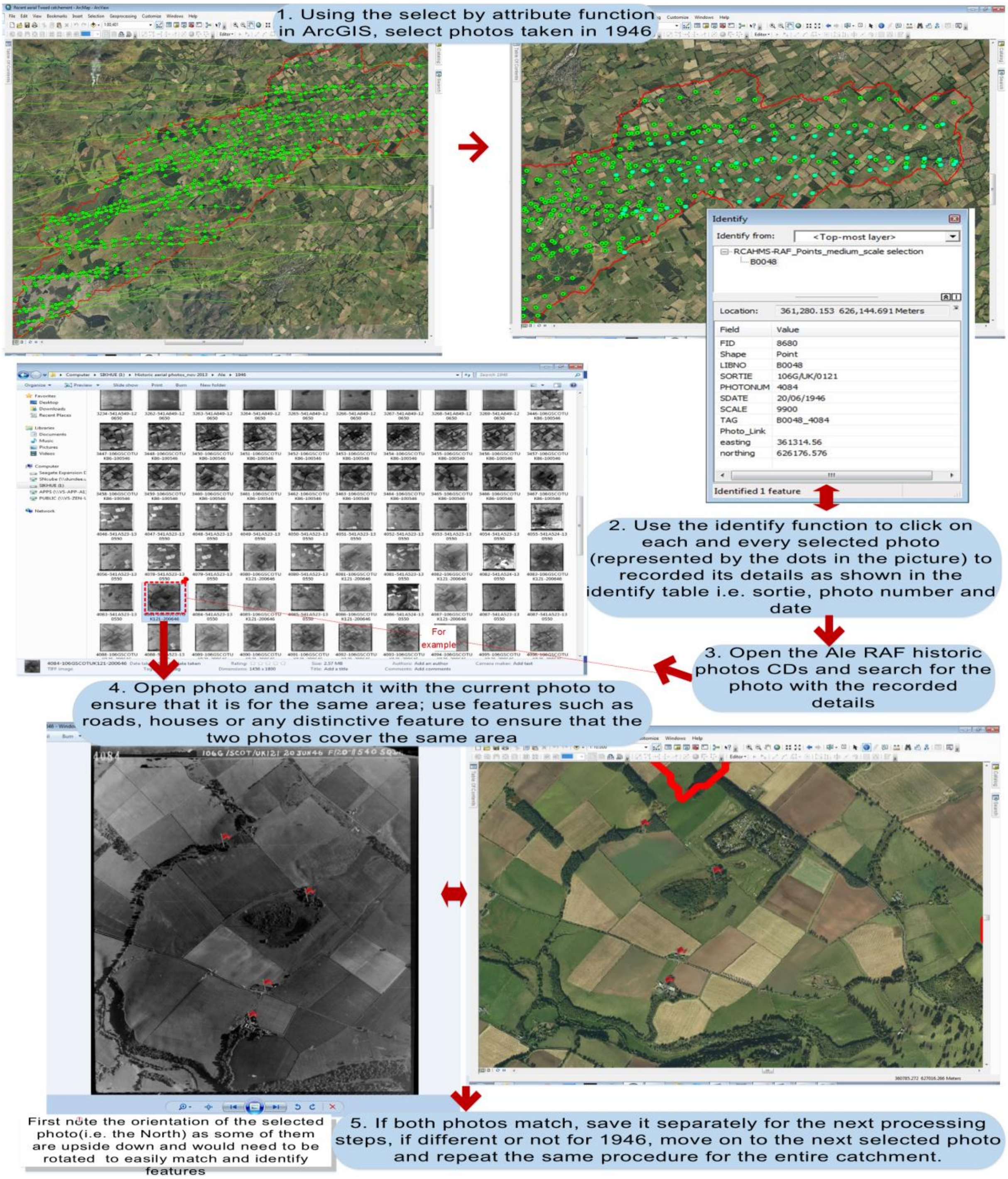


Figure 3-10: Selection process of historic air photos for the Ale sub catchment

Eddleston catchment

After searching the initial collection of the air photos in the CDs for the Scottish Borders as was done for the Ale catchment, it was discovered that the air photos that covered the Eddleston catchment were not there. As an alternative, an online search for these on the Royal Commission on the Ancient and Historic Monuments of Scotland (RCAHMS) website was made (i.e. national archive for historic aerial photographs in Scotland). Findings from the search showed that the RCAHMS had a collection of historic air photos that covered this catchment (figure 3-11). Such a collection had air photos ranging from the 1940s onwards but, as in the Ale catchment these were randomly clustered.

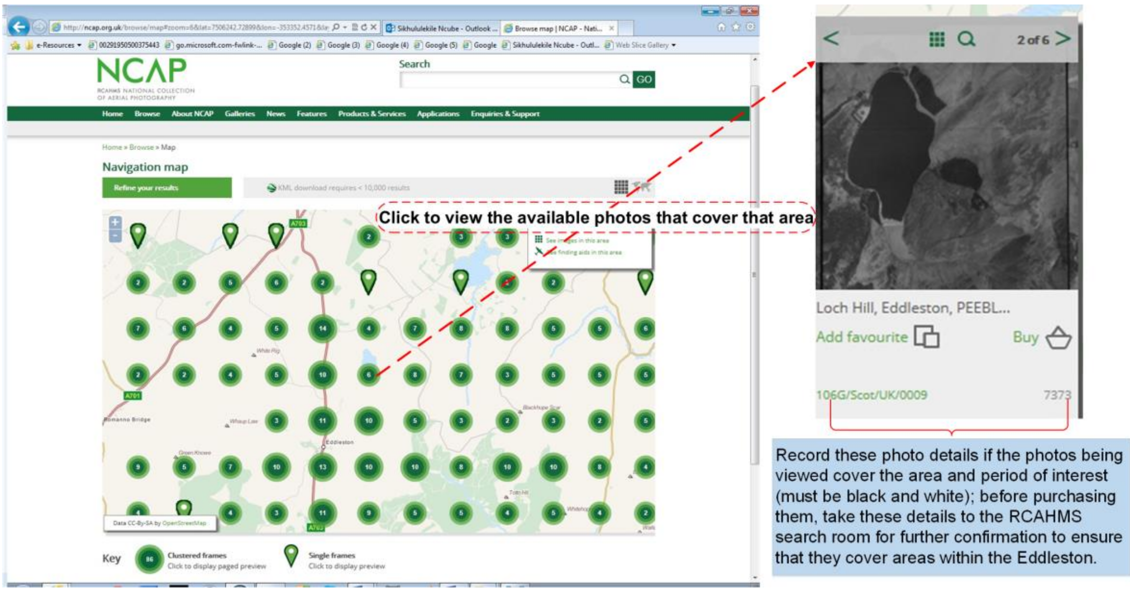


Figure 3-11: Online search for air photos for the Eddleston catchment

As illustrated in the figure above, the encircled numbered green dots show the number of available air photos that capture particular sections within the Eddleston catchment. The zoomed out picture in the figure above was selected from one of the green dots which consisted of air photos captured at different time periods some of which were acquired in the recent past. Since the zoomed out picture was captured during the 1940s, it met the required criteria and hence details of such photos were recorded.

After the online search, a visit was made to the RCAHMS aerial photography search room to verify the recorded photos and search for more air photos so as to get a complete coverage of the catchment. With the assistance of RCAHMS personnel, air photos from the 1940s that covered this catchment were then selected and purchased. Reference to the sortie plots showing flight paths that covered the Eddleston was also made. As in the case of the Ale catchment, the flight paths also reflected an unsystematic capture of these air photos during the RAF surveys (refer to appendix 3). In total, 132 digitally scanned unregistered air photos (scale 1: 10 000) were purchased for use in this study.

3.3.2 Reconstruction of the catchments landscape photo mosaics: use of Photoscan software

In order to avoid the time demands, expertise and labour intensiveness of using a stereoscope to interpret the historic air photos, a software (Photoscan Professional Edition Educational License from Agisoft LLC) was used to co-register[3] and stitch these photos together to produce high resolution orthophotos[4] (photo mosaics[5]). Using this software enabled the generation of these orthophotos in a format that can be handled, analysed and interpreted within GIS. This was of key importance in this study as the subsequent stages of data collection and analysis were done in GIS.

Photoscan is a photogrammetric software that uses a technique referred to as the Structure from Motion (SFM) from the computer vision research field (Verhoeven, 2011). The SFM technique is based on algorithms that detect and describe local features for each image and subsequently match those 2D points throughout the multiple images (Ducke et al., 2011, Verhoeven et al., 2012a). The output from this process is a representation of the structure/geometry of the landscape/scene captured (Morgenroth and Gomez, 2013).

Photoscan has a semi-automated workflow which can process all photos without knowledge of camera parameters. This means that photos that were randomly acquired over different time scales, using different cameras without ground control points can be processed to generate orthophotos (Lo Brutto and Meli, 2012, Verhoeven et al., 2012, Morgenroth and Gomez, 2013). This was a key consideration in this study given the random nature and absence of camera details for the historic air photos that were used.

[3] This refers to conversion of air photos to a common projection and coordinate system

[4] This is a photomap, which can be interpreted like photographs and which contain all the information content of the original photograph. They have one scale just like maps and can be overlaid with other data in GIS (Welch and Jordan, 1996)

[5] Used interchangeably with orthophotos in this thesis

Developing the orthophotos (photo mosaics) in Photoscan: the procedure

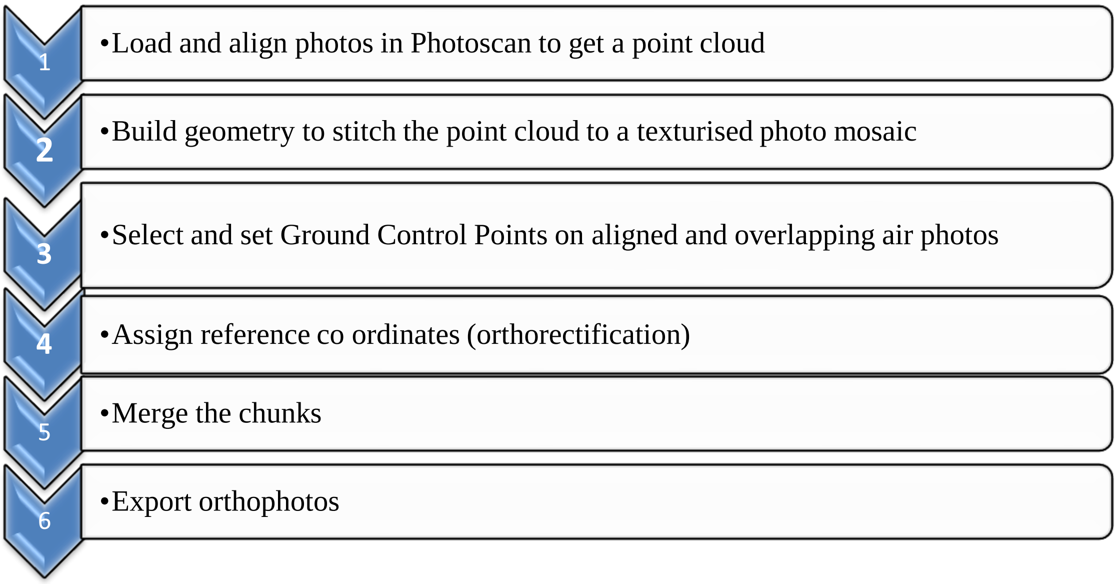


Figure 3-12: Overview of the processing steps in Photoscan[6]

Step 1: Loading and align the air photos

For the ease of processing, the photos were first grouped according to the sortie codes which matched their flight path and the date they were taken in 1946. They were then loaded into Photoscan as separate chunks which were later merged (refer to screen shot figure 3-13).

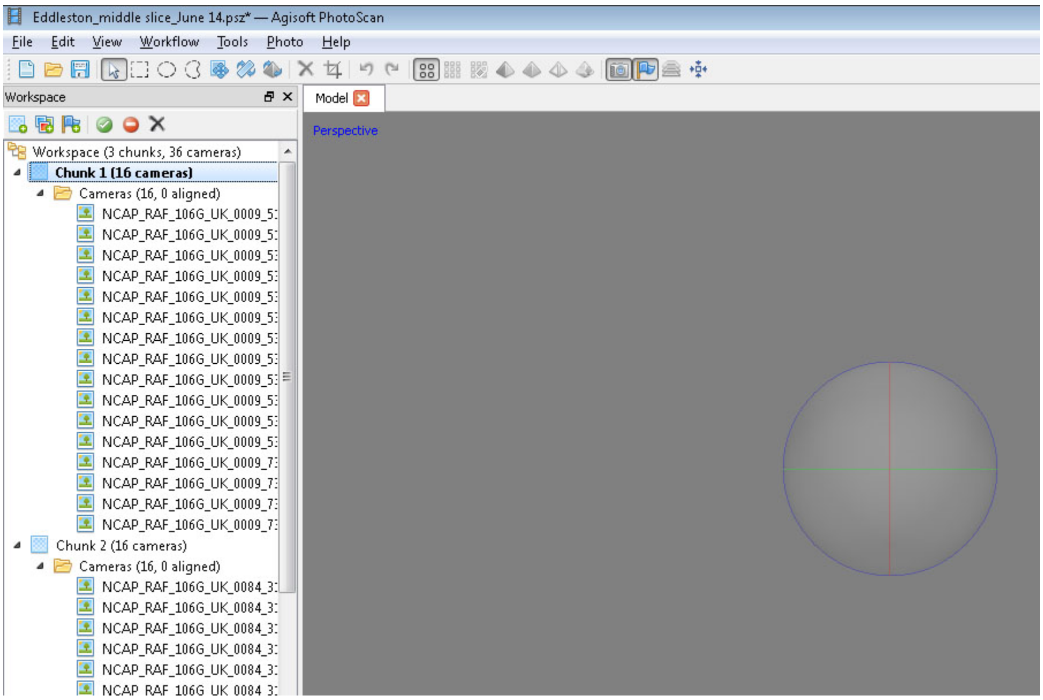


Figure 3-13: Load photos into the Photoscan workspace

[6] The same processing steps were done for both catchments.

The figure above shows the Photoscan workspace which allows for 3D viewing of the model during its development while the left pane shows the photos that were split into small sizeable chunks and uploaded in Photoscan.

Next, the align photos command from the Photoscan workflow menu was selected. In this process, the software searches for common points on the loaded photos and matches them, including the camera positions for each photo loaded. It reconstructs the positions from where the photos were taken (Screen shot figure 3-14).

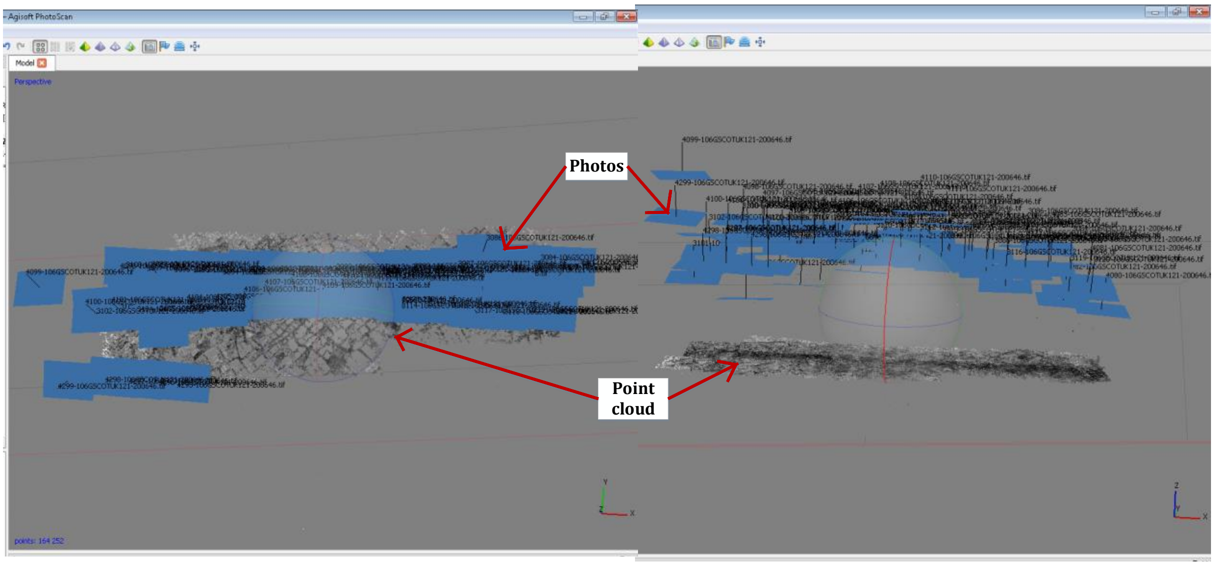


Figure 3-14: Reconstructed photo positions and point cloud

As shown in the figure above, the output from this alignment step was a point cloud[7] and camera positions from where the photos were taken (i.e. the blue patches in the figure). The point cloud corresponded with the varying positions from where the photos were taken including those taken at a tilt. For those photos taken at a tilt, the software only shows the camera positions from where they were captured, but it does not align them with the rest of overlapping photos and hence these are referred to as 'stray' or 'outliers' (figure 3-15).

[7] NB: The point cloud shown is an extract from one of the catchments and is being used to explain the procedures done. This will be done throughout the chapter but, note that the same procedure was done for the other catchment as well.

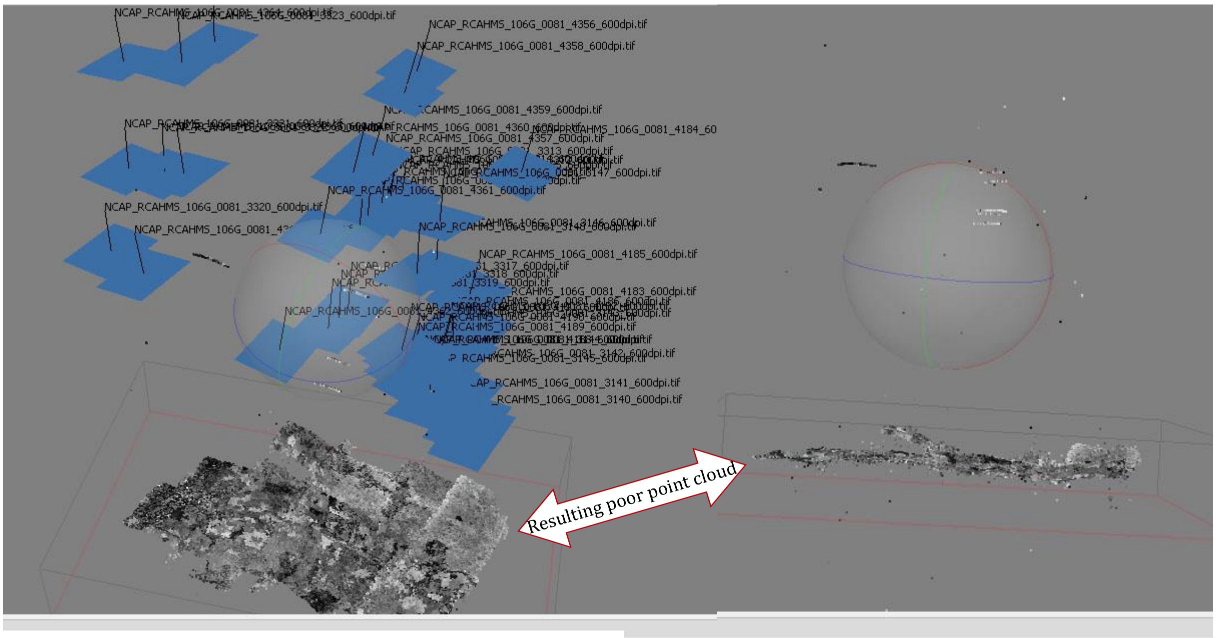


Figure 3-15: Example of stray photos

As illustrated in the figure above, outliers result in a scattered point cloud and these distort the reconstruction of the landscape structure. They had to be removed/edited so that a neatly, tightly spaced point cloud be processed further. This had to be repeatedly done to get a better alignment of all the photos in the chunks to ensure that the number of photos removed as outliers was minimised. In some chunks a large number of photos were not aligned and this was solved by creating a separate chunk/folder for these photos and aligning them again. A higher number of outliers meant a reduction in the areal extent of the catchments that would be reconstructed. Aligning of the photos with at least 60% overlap was also crucial for the following processing steps. Figure 3-16 shows examples of neat, even point clouds that were suitable for further processing.

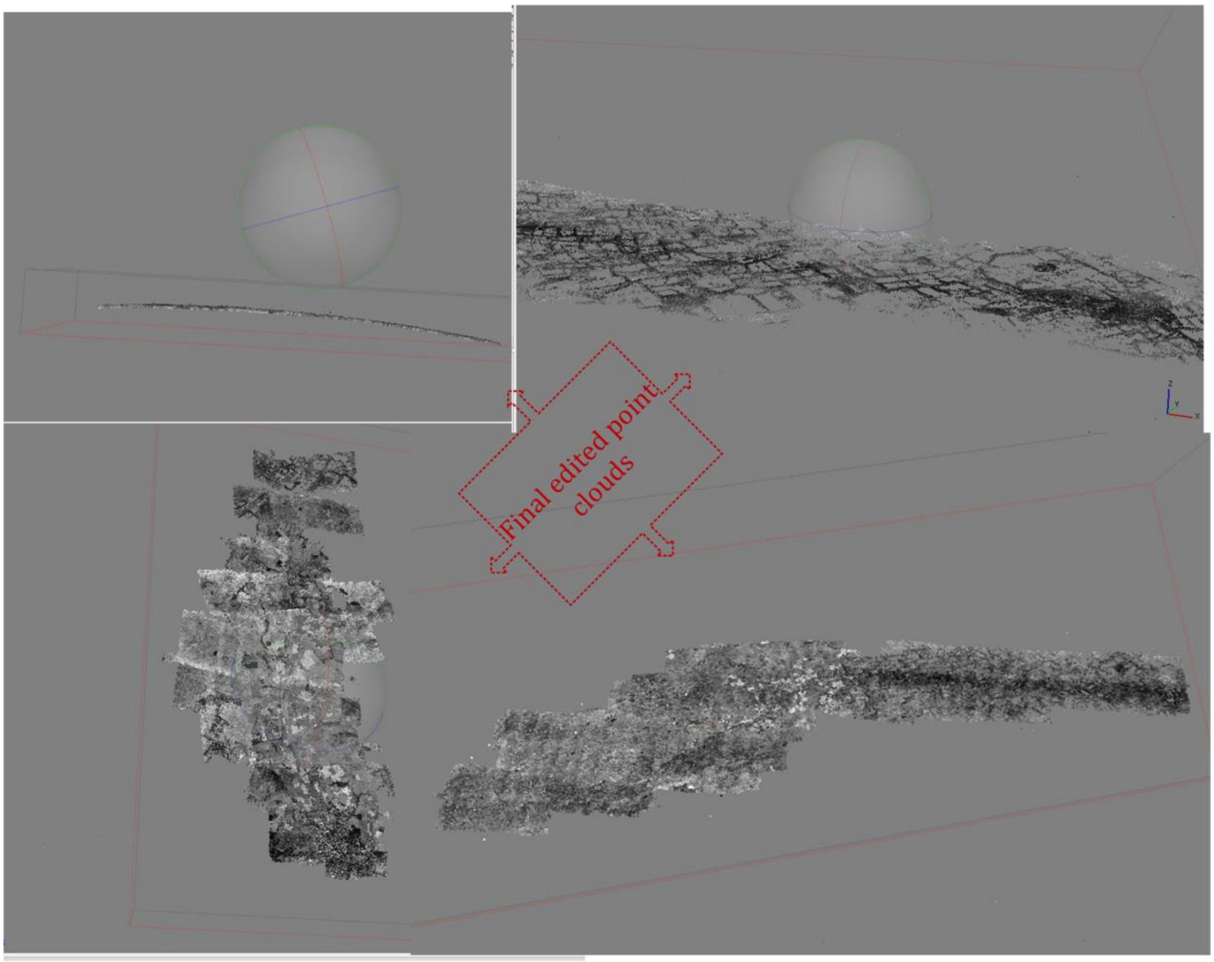


Figure 3-16: Examples of neatly aligned point clouds

Step 2: Building geometry

To do this, the point cloud had to be oriented within the software set reconstruction plane (boundary box). In this step, the software stitches (builds texture) the point clouds from the previous step into a texturized output (Refer to figure 3-17).

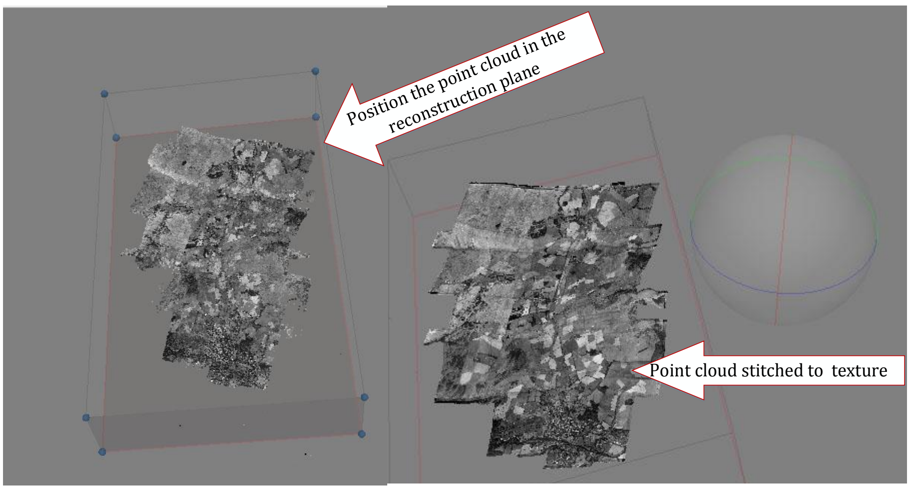


Figure 3-17: Illustration on building texture

Step 3: Selecting and Setting ground control points (GCPs)

In order to establish the exact spatial position and orientation of the photo mosaics developed in the preceding steps, ground control points (markers[8]) had to be placed on the air photos. To do this, all overlapping and aligned photos in step 1 were listed and their details recorded (number, date and sortie code). These were the photos on which the GCPs were to be placed. GCPs were placed on easily identifiable features with sharp edges whose locations could be pinpointed, like road intersections, field edges/corners, hills edges, woodland edges, reservoir edges, bridges, houses etc.

Prior to placing the GCPs in Photoscan, these same features and GCPs positions had to be also located on the base layers in ArcGIS so that their coordinate positions (latitude(x), longitude (y)) and altitude (z value) would be recorded (figure 3-18). The x,y values were recorded from current (2009) aerial photography, while the z value was recorded from the DTM. The Ordinance Survey topographic map was used to locate place names within the catchments.

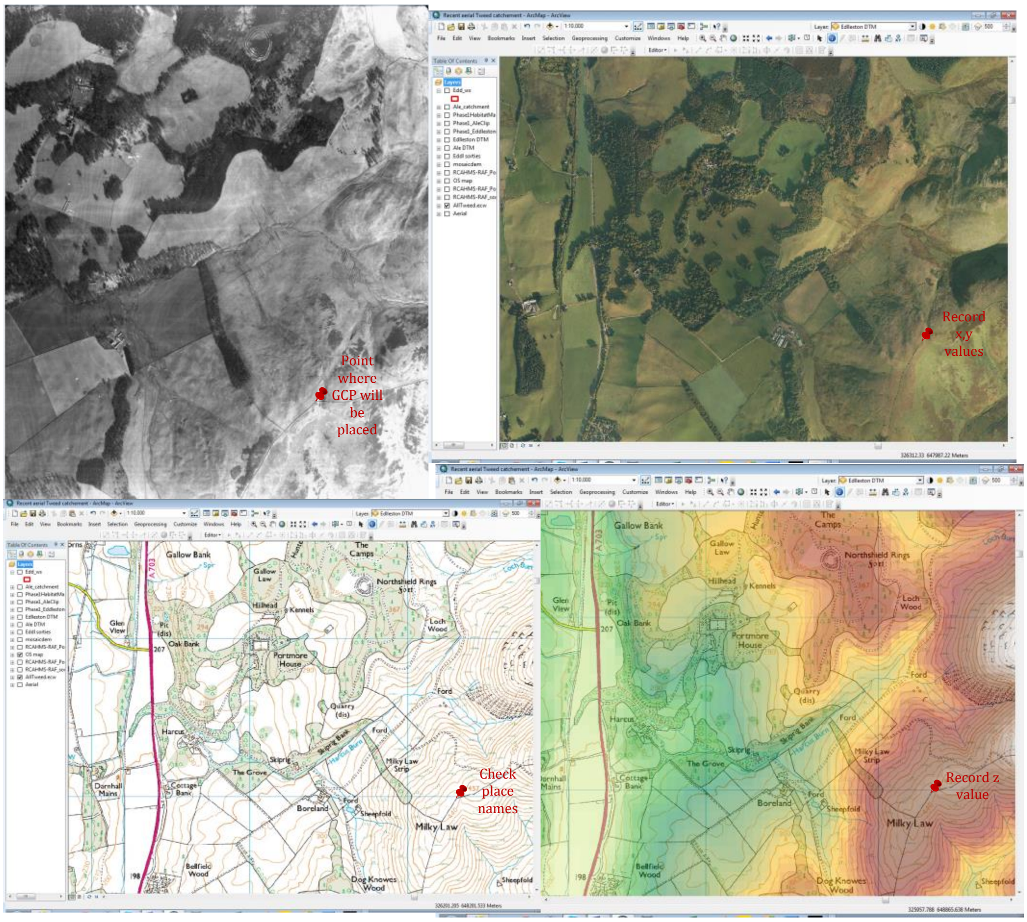


Figure 3-18: Illustration on selecting ground control points

Contains OS data © Crown Copyright 2015

[8] Used interchangeably with GCPs

As illustrated in the figure above, the process of recording the x,y,z values had to be done by cross checking and matching suitable GCPs positions between historic and current photos. If a suitable GCP position was identified, its location description and the x,y,z, values were recorded. Accurate recording of these values in ArcGIS was crucial as it influenced the accuracy of the orthophotos. Recording was done for a number of GCPs in order to have a wide distribution of these GCPs across the whole photo mosaics so as to provide a stable warp of the photo mosaic. More than 30 GCPs were selected for each catchments. Hughes et al. (2006) and Verhoeven et al. (2012a) advise that using more GCPs improves the accuracy of orthophotos and that is why more than 30 GCPs were selected for each of the study catchments.

After recording the location details of the selected GCPs in ArcGIS, these then had to be manually placed on the listed overlapping photos in Photoscan. To do this, the add makers/GCPs command was selected from the Photoscan workspace panel. Markers were then placed on the same point on overlapping photos. So, for example, if a field was selected as a suitable position for placing a GCP, a marker was placed on the field corner in one photo and the same marker had to be placed on the same field corner in the other overlapping photo (refer to figure 3-19).

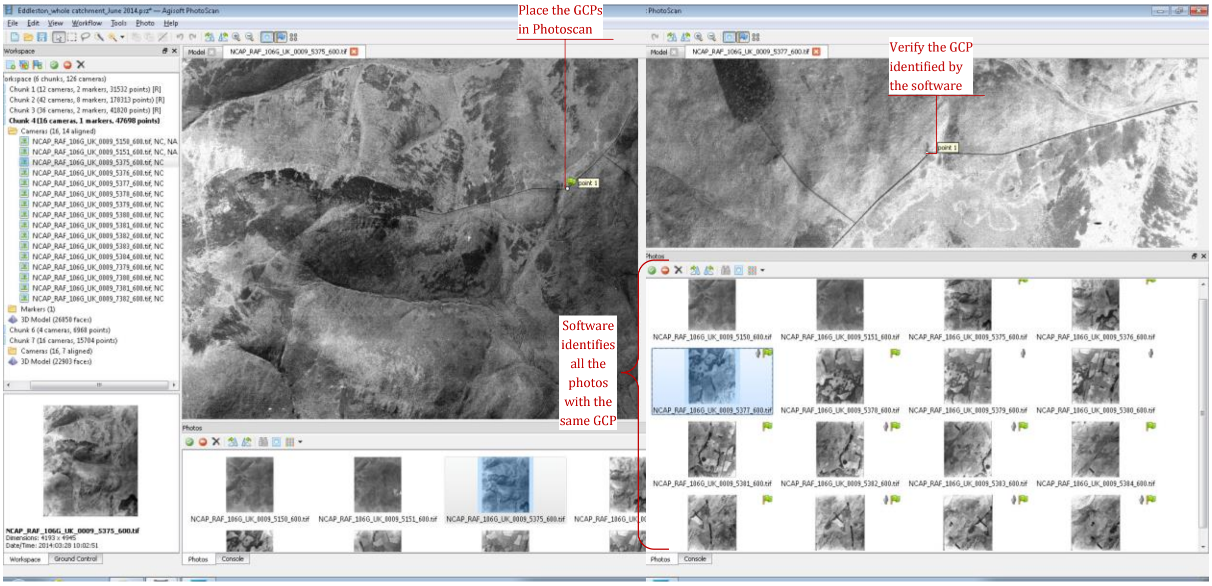


Figure 3-19: Placing of GCPs on overlapping air photos

As illustrated in the figure above, the markers had to be placed on at least two overlapping photos. However, in some cases there were more than two overlapping photos with the same selected feature on which the marker was placed. In such cases, the software automatically identified and placed markers on the rest of the photos with the same feature (as shown on the bottom right corner of the figure above. Such automatic placing of these markers had to be verified and in some cases edited so that they were positioned correctly (refer to top right corner of figure 3-19). The figure below shows how the GCPs were distributed in the Ale catchment after the completion of this step.

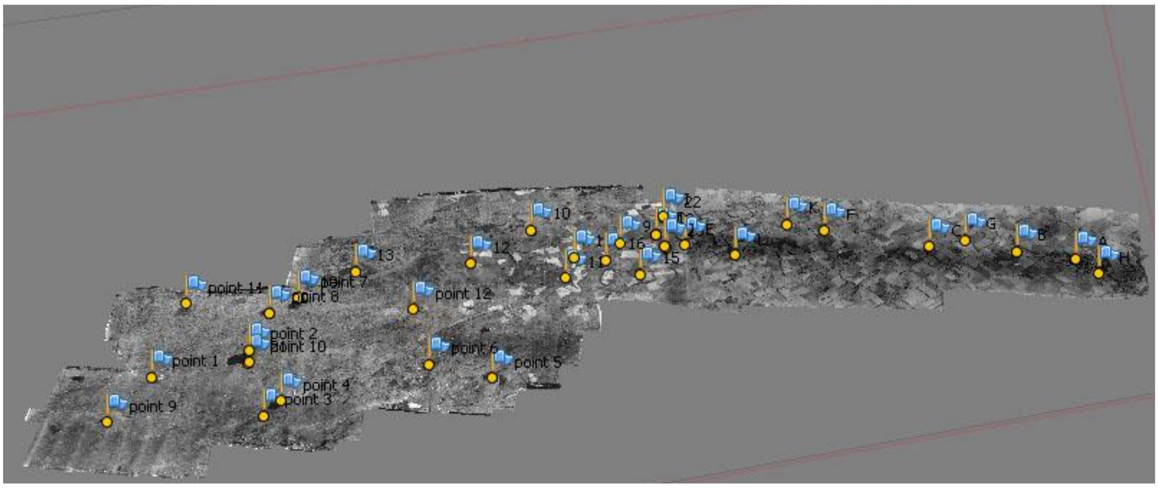


Figure 3-20: Distribution of GCPs within the Ale catchment

Step 4: Assigning reference coordinates (orthorectification)

After placing the markers on the photo mosaics, the Photoscan ground control panel was then populated with the x, y, z values for all the markers (figure 3-21). These were the values recorded in ArcGIS in the previous step. The process of placing these x, y, z reference coordinates is referred to as orthorectification (Lillesand et al., 2004). After capturing these values, the transformation settings were updated to the British National Grid co-ordinate system and the photo mosaics/ orthophotos were shifted/warped to their real world positions.

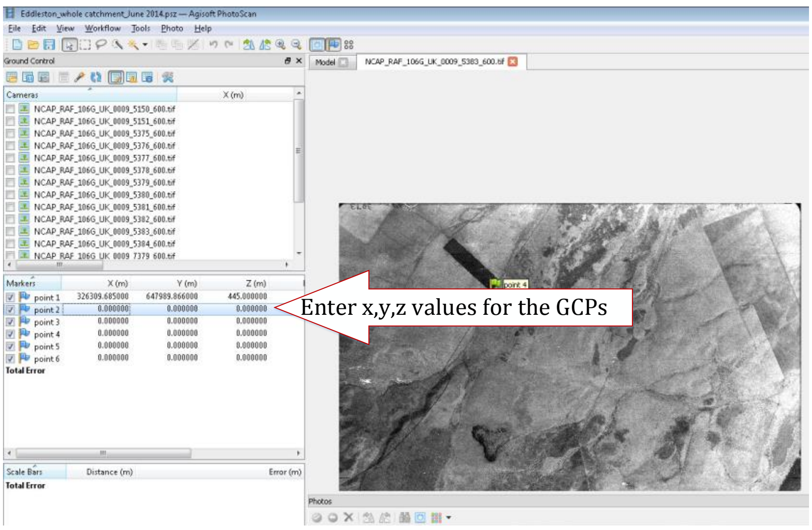


Figure 3-21: Entering x,y,z values for all the GCPs

Step 5: Merge the chunks

In this step, all the smaller chunks formed in step 1 were merged to get a photo mosaic/orthophoto for the entire catchment. This was done by selecting the “merge chunks” function from the software workflow command. The software relies on the GCPs/markers to merge the chunks.

Step 6: Export orthophotos

The sequential steps described above, resulted in the production of high resolution black and white orthophotos for the Ale and Eddleston catchments (presented in Appendix 8 and 9). These were imported into ArcGIS where air photo interpretation and habitat mapping was done (Stage 2).

3.4 Stage 2: Visual air photo interpretation and historic habitat mapping

3.4.1 Phase 1 habitat classification adopted for visual onscreen aerial photo interpretation

The Phase 1 habitat classification system is widely used in the UK as a standard approach for habitat classification (Cherrill and McClean, 1999b, Stevens et al., 2004, Jackson, 2000, Joint Nature Conservation Committee, 2010). It has been applied as a standard habitat classification method by environmental consultants, wildlife trusts, local authorities, national park authorities and the governmental conservation bodies. In Wales, for example, over 80% of the land surface is reported to have been mapped by the Countryside Council for Wales using the Phase 1 survey and classification system. It is also noted that over 70% of the land surface in the UK has been mapped using the Phase 1 method (Cherrill and McClean, 1999a, Cherrill and McClean, 1995). The Joint Nature Conservation Committee maintains this habitat classification standard across the UK (Joint Nature Conservation Committee, 2014).

The Phase 1 method was chosen for use in this study because the current habitat map (2009), which this study uses for comparison and habitat change analysis, was produced using this method. Thus it was considered appropriate to use the same method to produce the historic (1946) habitat map so as to reduce errors and uncertainties resulting from using different methods. Cherrill and McClean (1995) caution that habitat data obtained using different methodologies and differring habitat classifications could increase inaccuracies and errors in comparing such data. Stevens et al. (2004) also advise that reliable information can be obtained by analysing comparable sets of habitat data collected using similar methods and based on the similar habitat classification system.

The principal Phase 1 survey technique is a field based ground survey, but this technique is criticised for being costly both in terms of time and labour (Groom et al., 2006, Lucas et al., 2007, Juel et al., 2013). In addition, the integration of results from the Phase 1 field survey technique with other datasets can be cumbersome given the hard paper format of the collected map data (Cherrill and McClean, 1995). On this basis, the use of remotely sensed data such as aerial photography, as used in this study, and satellite imagery has been suggested as suitable alternatives to field based habitat mapping techniques (Cherrill et al., 1995, Juel et al., 2013). Remotely sensed data sources offer an added advantage of providing habitat data in digital formats which can be combined and easily analysed with other datasets in Geographical Information Systems (GIS).

The use of satellite imagery was not pursued in this study because the two main types of satellite imagery (Landsat TM and SPOT system) recognised as suitable for Phase 1 habitat mapping have other limitations. Landsat imagery cannot identify the full range of Phase 1 habitat classes due to its coarse spatial resolution (approximately 30m spatial resolution), while the spectral resolution of the SPOT system is reported to be poor (Joint Nature Conservation Committee, 2010, Lucas et al., 2011, Hooftman and Bullock, 2012). Hooftman and Bullock (2012) for example, observed that some broad habitat types like houses, gardens, roads, inland water features were not well represented in their study due to the coarse resolution of Landsat imagery. Even though satellite imagery with very high spatial resolution like Quick bird/IKONOS could have been an alternative, these are significantly expensive (Groom et al., 2006). Furthermore, satellite imagery availability only dates back to the 1970s while the time period which this study focusses on goes further back than this and aerial photography provided the desired longer ecological dataset.

Phase 1 habitat classes

The definition of habitat classes/types in the Phase 1 habitat system is primarily based on dominant and characteristic vegetation species (Joint Nature Conservation Committee, 2010). In cases where vegetation is not the dominant component of the habitat, topographic, soil, land use characteristics and other substrate features are used to define the habitat classes (Cherrill and McClean, 1999a, Cherrill and McClean, 1995, Joint Nature Conservation Committee, 2010). The Phase 1 handbook manual (published in 1990 and reprinted in 2010) provides procedures for standardised classifying and mapping habitats. The handbook also gives detailed descriptions of Phase 1 habitat classes and the associated codes. The habitat classes are split into divisions and within each major division there are sub divisions which are also further sub divided to give a hierarchical structure to the classification (refer to appendix 4). For example, woodland is sub divided into broadleaved, coniferous and mixed woodlands. Each of these divisions is further divided into semi-natural and plantation habitat classes.

3.4.2 Visual manual interpretation of aerial photographs

Manual interpretation of aerial photographs using a stereoscope has long been criticised for being difficult, as well as time consuming and labour intensive (Taylor et al., 2000, Lillesand et al., 2004, Morgan et al., 2010). It also needs skilled and experienced personnel. Manual aerial photography interpretation has also been criticized for being subjective and prone to inconsistencies and errors. These criticisms have driven some researchers (Morgan et al., 2010) to advocate for the use of automated digital analysis techniques associated with satellite based land cover mapping such as pixel based and object based classification systems.

Despite the availability of these automated digital analysis techniques, this study was based on visual interpretation of aerial photographs. This was opted for over these newer approaches due to a number of reasons. Firstly, since this is a historic study, the main data source being interpreted were black and white aerial photographs which have limited spectral band/values as these only have one band with shades of gray. It would have been challenging and possibly mis leading for automated classification systems to consistently distinguish the different habitat classes based on limited spectral information. This would also have required extensive manual quality control checks and editing, as was encountered by Jarman et al. (2010). Automated classification stystems have proved to produce better results on imagery with high spectral variation (Jarman et al., 2010).

Secondly, automatic classification systems could not be applied owing to the quality of some of the photos used. These variations in photo quality were influenced by variations in look angle and illumination across the flight paths when the photos were captured. Feature extraction could have been a challenge if for example, an object based classification was to be applied using varying photo qualities.

Thirdly, significant amount of development work and training data is required for an automated classification (Morgan et al., 2010) and this could have been a challenge in terms of both time and costs to access and prepare such data. There was not enough information for either an object or pixel based classification approach. Instead, with the data that could be accessed, and the time and resources available, on screen visual interpretation of the aerial photographs was more feasible.

It should be noted that this study does differ from and improve on traditional manual air photo interpretation of a pair of overlapping photos in a stereoscope. Current technological improvements (photoscan software) were used allowing for the production of GIS compatible photo mosaics which can be visually interpreted on screen aided with ancillary datasets while also digitising changed areas.

3.4.3 Basic characteristics in aerial photography interpretation

Morgan et al. (2010), following Lillesand et al. (2004) explain that aerial photography interpretation is about the recognition, identification and significance of features or objects on photographs. Air photo interpretation is also accompanied by map editing, feature delineation, annotation and classification of polygons. In order to facilitate this interpretation process, there are basic characteristics that are used to identify and classify features in aerial photography. The table below, gives a description of these basic image interpretation characteristics as explained by Morgan et al. (2010) and Lillesand et al. (2004).

Table 3-1: Basic characteristics in aerial photography interpretation

Characteristic	Description
Shape	This refers to the outline of individual features or objects e.g. crown shape of trees. E.g. coniferous trees have pointed tops, cone shaped and have a darker tone.
Pattern	This relates to the spatial arrangement of objects e.g. the orderly nature of woodland plantations as opposed to scattered semi-natural woodland or scrub.
Size	This relates to the relative and absolute size of features or objects on the photography. This is particularly useful in understanding ecological patterns e.g. sizes of individual trees, relative sizes of habitat patches can have various ecological implications.
Tone (grayscale variation)	Defines the brightness and spread of colour on a photo. E.g. smooth spread of gray for improved grassland and the dark tone for cultivated land in a black and white photograph or a water body has a very dark image tone
Texture	Refers to the frequency of tonal change in an image and determines the visual smoothness or coarseness/roughness of features in the image.
Shadows	Show the shape or outline/silhouette of the certain features/objects e.g. shadow cast by trees and scrub will be of differing heights.
Site	Refers to the topographic location of features e.g. some habitat types are expected to occur uplands e.g. acid grasslands while others habitat types like riparian woodland would be mainly expected to be located on lowland sites.
Association	Pertains to the existence and occurrence of certain features or habitat types in relation to others e.g. heath would be easy to identify uplands where there are bogs and acid grassland as these are known mosaics.

3.4.4 Air photo interpretation characteristics used to identify broad Phase 1 habitat classes

Prior to air photo interpretation, field familiarisation visits were done within the study areas to understand the landscape and identify the main habitat types. This was also supplemented by literature review (i.e. reports, documents, journals) about the study areas and the Tweed catchment as a whole. Such knowledge and background information was useful during the air photo interpretation phase. For example, knowing that the lower part of the Ale catchment is mainly under arable farming meant that the dominant and expected habitat classes could be hypothesized. Equally, there were habitat types which were expected and known to be commonly found in the uplands. There were also some habitat classes that were expected to be found throughout the catchment and others not likely to be present at all. Such background information was used to inform the interpretation and assigning of respective habitat classes.

The first part of the interpretation and training was done together with an ecologist from Environment Systems Ltd (Dr Katie Medcalf) experienced in air photo interpretation, habitat and ecosystem services mapping. Meetings with her also occurred during the interpretation process for quality control, to ensure consistency and agreement on confusing habitat classes.

Since aerial photography interpretation was done on black and white orthophotos, differing shades of gray were mainly used to distinguish, identify and differentiate habitat classes. Attributes, appearance/shape of features and characteristics described above formed part of the criteria used to assign habitat classes. The high spatial resolution of the orthophotos (scale 1: 10 000) made it possible to distinguish features so that it was possible, for example to interpret whether a woodland type was broadleaved, coniferous or mixed based on the shape of the tree crowns and height and shape of the shadow patterns.

Other indicators such as management practice e.g. evidence of plough lines and the physical environment i.e. topography were also used to interpret the shades of gray. This was supported by ancillary data sets (Appendix 5). This supplementary information e.g. from the hill shade layer, the elevation, slope and convexity of slope assisted in the identification and classification of habitats in the uplands section of the catchment. Importantly, the current habitat map and the current colour air photos were the main guiding datasets as the colour aerial photography allowed for better visualisation of habitats. Of great benefit in using the aerial photographs was that some features e.g. reservoirs, streams, buildings, plantations that were present back then and are still present now were easily recognised.

Assigning of respective habitat class names was according to the categories in the Phase 1 habitat handbook (refer to appendix 4 for a detailed list of these classes). Below is a description of the main attributes and characteristics that were used to identify the broad phase 1 habitat classes present in the study areas, based on the Phase 1 habitat guidelines, basic characteristics in air photo interpretation (table 3-1), guidance from the experienced ecologist and experience from the field visits.

A. Woodland and scrub

Identification of the main **woodland** classes i.e. broadleaved, coniferous or mixed was based on the shape of the tree crowns, shadow patterns and height. Broadleaved woodland has an almost round/circular shaped crown. Coniferous woodland plantations were identified as large plantations with well-defined edges, dark photo tone, cone shaped crowns, orderly pattern of the plantations, trees appeared to be of even age and height in regular rows. The evidence of furrow patterns or faint parallel lines in close proximity with coniferous woodland plantations was used to identify recently planted or recently felled woodland. Mixed woodlands had a mix of various crown shapes. In terms of the anticipated location/site within the catchments, woodlands were expected to be found anywhere but with extensive coniferous plantations in the uplands.

Scrub appeared to be more irregularly dense or of scattered distribution, forming different patterns compared to the woodland plantations. It was also mainly distinguished from the woodlands by the shorter height of their shadows and small size compared to woodland. In terms of site and association, scrub was expected to be found anywhere within the catchments. In the uplands, for example, it was expected to be found in association with other habitat types forming mosaics.

Broadleaved parklands were identified by the presence of big trees scattered within enclosed managed grassland park areas. These were mainly found close to farm houses or village centres. Likewise, amenity improved grassland areas such as golf courses and playing fields were identified by the presence of expansive coverage of improved grassland surrounded by big trees. Reference to the ancillary datasets, especially the current aerial photography and the OS map, assisted in the identification of names of such parks and golf courses.

B. Grassland and marsh

Grasslands are associated with areas of relatively level or gently sloping terrain and the degree of agriculture improvement is noted to be a key factor in the classification of grasslands (Joint Nature Conservation Committee, 2010). Improved and poor semi-improved grasslands are those that have been influenced to a greater extent by drainage, application of fertilizers, grazing control and growing of grass species that are of agricultural use. Improved grassland was identified by the smooth even spread of the gray colour tone within enclosed fields. Poor semi-improved grassland was identified by the uneven spread and rough texture of the gray tone within enclosed fields. This was mainly found within farm estates and hence was mainly expected to be found in the low lying parts/arable areas of the catchments. However, parts of such areas were also identified in upland areas marked by the presence of field boundaries.

Semi-improved acid and neutral grassland are those that are in transition as they have been influenced by agricultural practices but they still retain the species that are characteristic of unimproved grassland (Joint Nature Conservation Committee, 2010). The photo texture for semi-improved acid grassland appears to be speckled with light and dark tones/spots and with assumed dominance of cotton grasses (*Eriophorum spp*). Unimproved grasslands are those that have not been sown, have low grazing or burning intensity and are indicated by an uneven photo texture. Acid grassland is noted to be mainly found unenclosed in the uplands either located at the bottom or inclined part of the slope, while neutral grasslands were on open land at the margins or borders of fields and along roads. Marsh/marshy grassland is mainly found where there are indications of waterlogging or wet areas (Lucas et al., 2007). Marshy grassland was also expected to be around areas of open water (reservoirs) and along rivers and streams.

C. Bracken

Bracken mainly occurs on steep slopes extending along valley sides and is characteristic of upland farms (The Macaulay Land Use Research Institute, 1993). It has different stages of growth and the time/season when the air photos were taken was the time when it was assumed to be green. Bracken was identified on the photos based on the lighter tone compared to the dwarf shrub heath.

D. Heathland

Bogs, blanket bogs and heath are usually found in larger continuous mosaics (Lucas et al., 2007). Heath, either dry or wet, was identified by dark patches on sloping sides in the uplands with dark tones/patches and an uneven photo texture. Dry dwarf shrub heath was marked by very dark patches and dense photo tones with evidence of harvesting or management control, marked by strip patterns (The Macaulay Land Use Research Institute, 1993). Dwarf shrub heath was expected to be located on hill slopes. Mosaics, especially acid grassland/heath were very common and covered extensive areas.

E. Mire

The degree of agriculture improvement is noted to be a key factor in the classification of bogs (Joint Nature Conservation Committee, 2010). Blanket bogs were expected to be found in the hills and uplands and were associated with shallow slopes. Using the DTM hill shade, current aerial photography and soil map layer, blanket bogs were identified by the edge effect of peat over concave or convex moderately sloping ground in the uplands. Blanket bogs or bogs with evidence of drainage lines were classified as modified bogs.

Fens and mires are associated with relatively level terrain. Flushes for example, occur on gently sloping ground and can be identified by a triangular pronged pattern close to streams and within valleys. A basin mire was identified by its location downslope while a valley mire was found on the floor of small valleys. These were also found in association with the standing water or swampy areas.

G. Open water

Reservoirs, lochs, rivers and streams were some of the features that were distinct from the air photos as they were shown by a very dark/almost black photo tone and distinctive shapes. Rivers and streams could also be identified by their meandering pattern. Rivers and streams were also identified by their linkage to reservoirs or lochs as they were either originating from or feeding into these standing water sources. Open water sources (reservoirs and lochs) were mainly located in the uplands. Other water sources, like small ponds, ditches were found anywhere within the catchments. All these open features were also present in the current aerial imagery and they were easily identified.

I. Rock exposure and waste

The main features identified under this category were the quarry sites and refuse tip/dumpsite. These could be seen on the air photos by the presence of open dug up pits and excavation sites.

J. Miscellaneous

This category included boundary features such as **hedgerows** (both stonewall and tree/bush hedges). These were mainly identified as dividing boundaries between enclosed fields, close to farm houses, along the roads. **Built up areas** i.e. farm houses, village centres, roads or any form of construction or highly artificial land cover types were also included in this category and these could be easily identified on the air photos. The currently existing roads are still at the same place as they were in 1946, however, some of these have been widened and the number of linking roads increased. Features such as roads were used as a guideline to locate places in the black and white orthophotos.

Cultivated/disturbed land-arable was identified mainly by the presence of enclosures such as fences, hedges and evidence of plough lines within fields. The picture tone was almost "white" in some fields. Such areas were expected to be dominant in the lower parts of the catchments, on generally flat terrain and within farm estates.

3.4.5 Air photo interpretation and historic habitat mapping: the procedure

3.4.5.1 Datasets used

The manual onscreen visual interpretation of the 1946 black and white orthophotos was done in ArcGIS. The current (2009) habitat map (vector format) was used as a guide in mapping the 1946 habitat map. The 2009 habitat map was essentially edited and backdated to 1946. The backdating approach used in this study is similar to studies done by Thomson et al. (2007) and Jauhiainen et al. (2007). Figure 3-22 lists the steps that were followed in constructing the 1946 habitat map.

The main data sets used to create the 1946 habitat map were the orthophotos from the stage 1, the current (2009) habitat map and the current (2009) colour aerial photography for the Ale and Eddleston catchments. These main data sources were supplemented with ancillary data sets which aided in orthophoto interpretation and habitat mapping. These included the OS topographic map, the DTM hill shade, the hedgerows and soil layers for the two catchments (refer to appendix 5).

These datasets showed landform, slope and hydrology and assisted in the differentiation of habitats based on their known location within the landscape. This was based on the biogeographical understanding of where different habitat types occur within the landscape. For example, there are some habitats which are known to only occur at certain altitudes in the British countryside, for instance acid grassland is mainly associated with the uplands. Thus the ancillary datasets were very instrumental in the interpretation process and ArcGIS allowed for switching on and off of these data layers for comparing and cross checking before assigning the probable habitat classes.

1. • Load datasets in ArcGIS
2. • Change habitat symbology to show the polygon boundaries/outline only and save the habitat layer with a different name e.g. Ale 1946 habitat map
3. • Zoom to polygon delineation (editing) and interpretation scale
4. • Open attribute table and add new column for 1946
5. • Start editor
6. • Cross check the land unit/area being interpreted with ancillary datasets to assign the probable 1946 habitat class and also refer to the image interpretation characteristics
7. • Edit polygons i.e. merge or split to trace out the 1946 habitat class and label if changed from the current, if not changed relabel using the interpreted current habitat class name
8. • Add the labelled habitat class to the 1946 column in the attribute table and save the edits
9. • Repeat steps 6-8 moving from one land unit to the other until the whole catchment is interpreted at the field level scale i.e. 1: 4000 and also do the same for the other catchment

Figure 3-22: Overview of the historic habitat mapping procedure

Step 1: Load and overlay datasets in ArcGIS

The first step was to load and overlay all the data set layers. The top most layer was the current (2009) habitat vector layer (map) which was edited to generate the 1946 habitat layer (map). The base map to it was the 1946 orthophoto (black and white) layer which was being interpreted and the current aerial photography. The figure[9] below shows the data layers overlaid in ArcGIS.

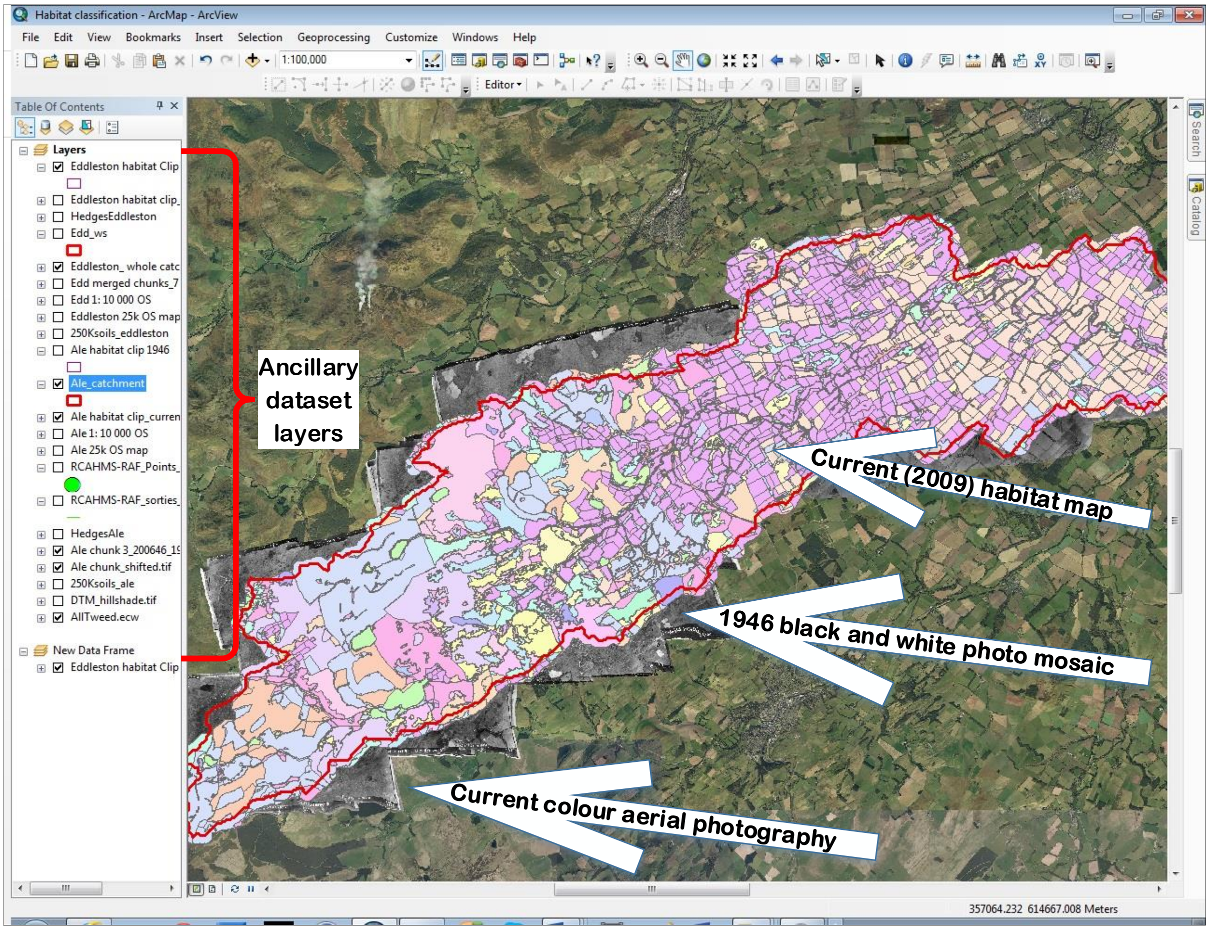


Figure 3-23: Dataset layers used for visual onscreen aerial photo interpretation

As shown in figure 3-23, the main data layers were accompanied by ancillary data set layers which were switched on and off during the interpretation. These were useful in deciding and assigning the habitat classes. For example, the DTM hill shade layer would show the topographic location of the habitat within the catchment while the soil layer showed the underlying soil type, and all this information assisted in assigning the different habitat classes.

[9] NB: the screen shot examples show either of the catchments but the same procedure was done for both catchments.

Step 2: Change habitat symbology

The habitat symbology was changed for the top most 2009 habitat layer (vector format) to show only the polygon outlines/boundaries (refer to figure 3-24). This was done so that the underlying 1946 orthophoto being interpreted could be clearly seen.

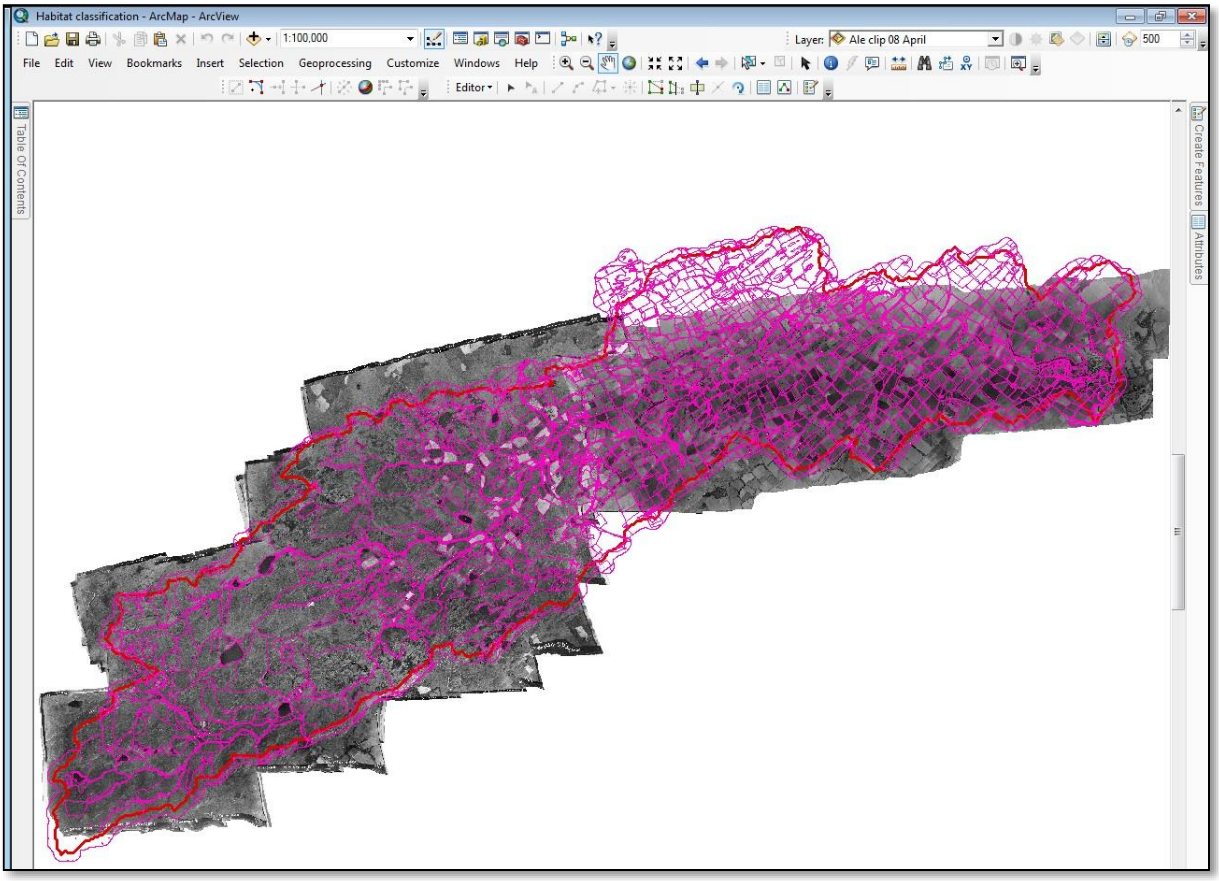


Figure 3-24: 2009 habitat map vector layer polygon boundaries for the Ale sub catchment laid over the 1946 orthophoto

As illustrated in the figure above, the polygon outlines/boundaries of the 2009 habitat layer could be easily seen against the black and white orthophoto base layer. The editing and relabelling of the 2009 habitat map was based on the interpreted orthophoto. The creation of the 1946 habitat layer thus capitalised on the existence of the mapped contemporary habitat layer covering the exact same areas. It would have been far more challenging to create a new vector layer while also interpreting the air photos. Instead, editing the current habitat layer to create the 1946 layer offered an advantage of retaining the pattern of polygons (Thomson et al., 2007).

Step 3: Zoom to polygon delineation and interpretation scale

The display scale of 1: 4000 as recommended by Joint Nature Conservation Committee (2010) was used for air photo interpretation. At this scale, different habitat types (using the recommended minimum mapping area of approximately 0.1ha) could be clearly delineated. For example, hedge rows could be distinguish and this was also attributed to the high spatial resolution of the air photos. Figure 3-25 below shows the zoomed in section from the Ale catchment.

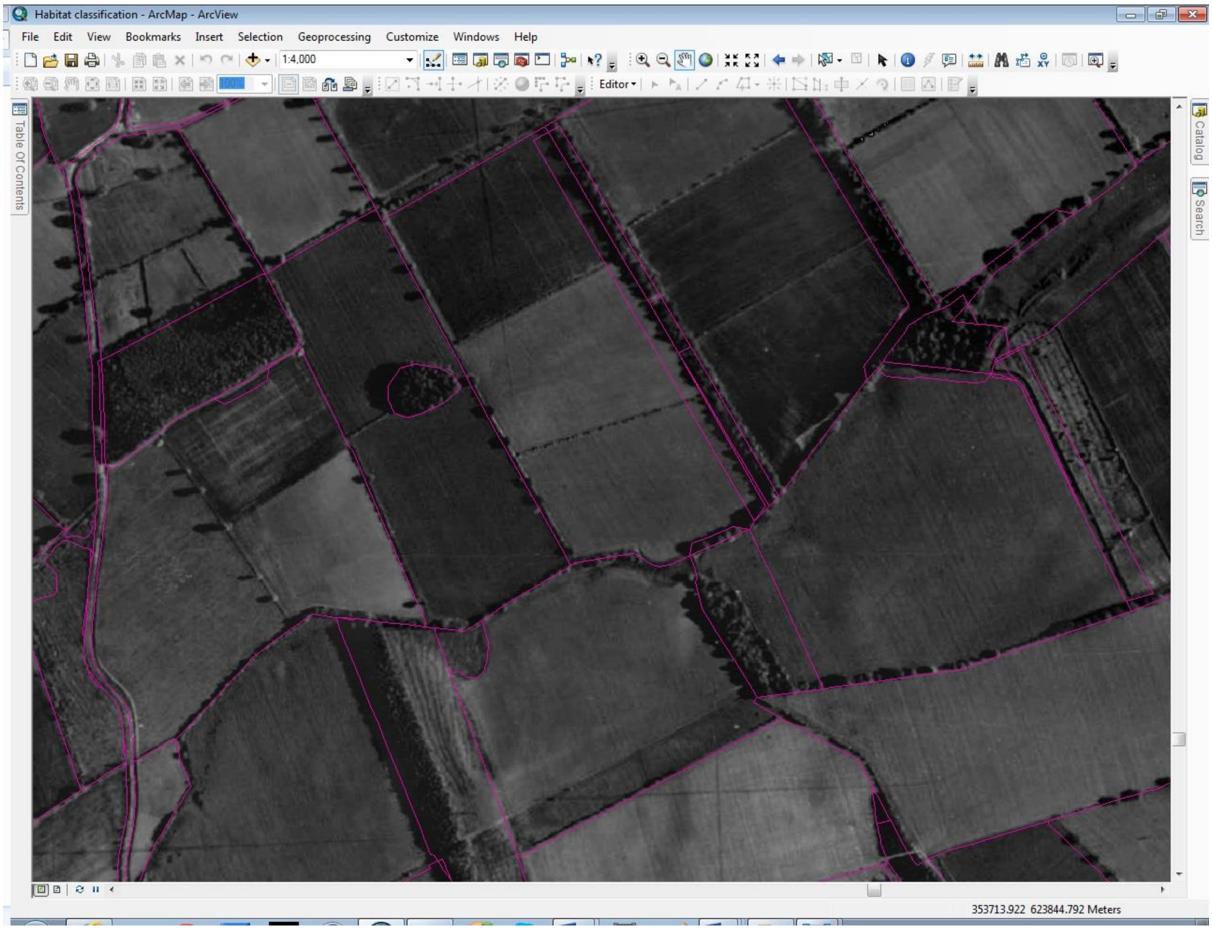


Figure 3-25: Figure: Interpretation scale 1: 4000

As shown in the figure above, at a display scale of 1: 4000, field boundaries could be easily demarcated. The polygon lines show the boundaries of these habitat classes in 2009 and as part of creating the 1946 habitat layer these boundaries were adjusted and edited to match the underlying orthophoto habitat boundaries. However, zooming in and out to various scales was done for clarification of habitat type and viewing the wider landscape and the context within which the habitats were located.

Given the sizes of these catchments, the interpretation was done in small chunks to ensure that the entire catchment was interpreted. Features such as roads, buildings, rivers were used as guidelines to ensure that a complete coverage of the catchments was achieved.

Step 4: Start air photo interpretation and labelling of habitat classes

As already discussed above, the actual air photo interpretation process involved identifying different habitat classes using the air photo interpretation criteria outlined earlier. Reference to the current aerial photo and ancillary data sets was also done before assigning the respective habitat classes. So for example, if an area in the current 2009 habitat map was labelled as a coniferous woodland plantation, the air photo was checked to see whether it was the same in 1946 or it had changed to another habitat type. If changed, the habitat type at that time would be labelled accordingly by using the editing function in ArcGIS to either clip or merge and label the 2009 habitat map vector polygons to match the interpreted 1946 habitat class boundaries. If a habitat class had not changed between these time periods e.g. reservoirs and running water, these were relabelled as they are.

Interpretation and editing the polygon boundaries was done simultaneously moving from one land parcel to another until the whole catchment was completed. The 1946 habitat map was gradually built alongside the interpretation and polygon editing process. The flow diagram (Figure 3-26) below is an illustration of this process. This process took four months, 3132 and 6789 polygons were edited for the Eddleston and Ale catchments respectively.

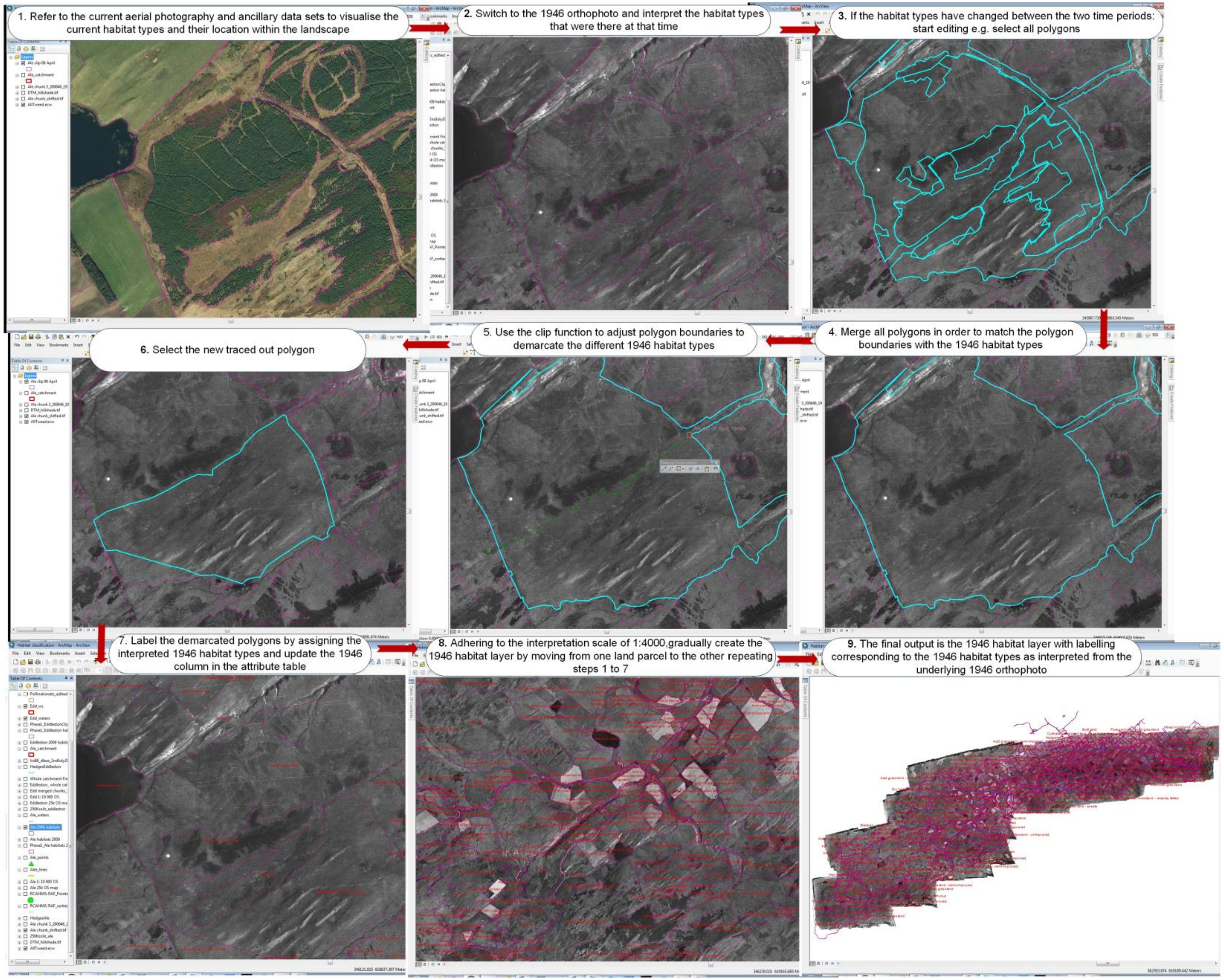
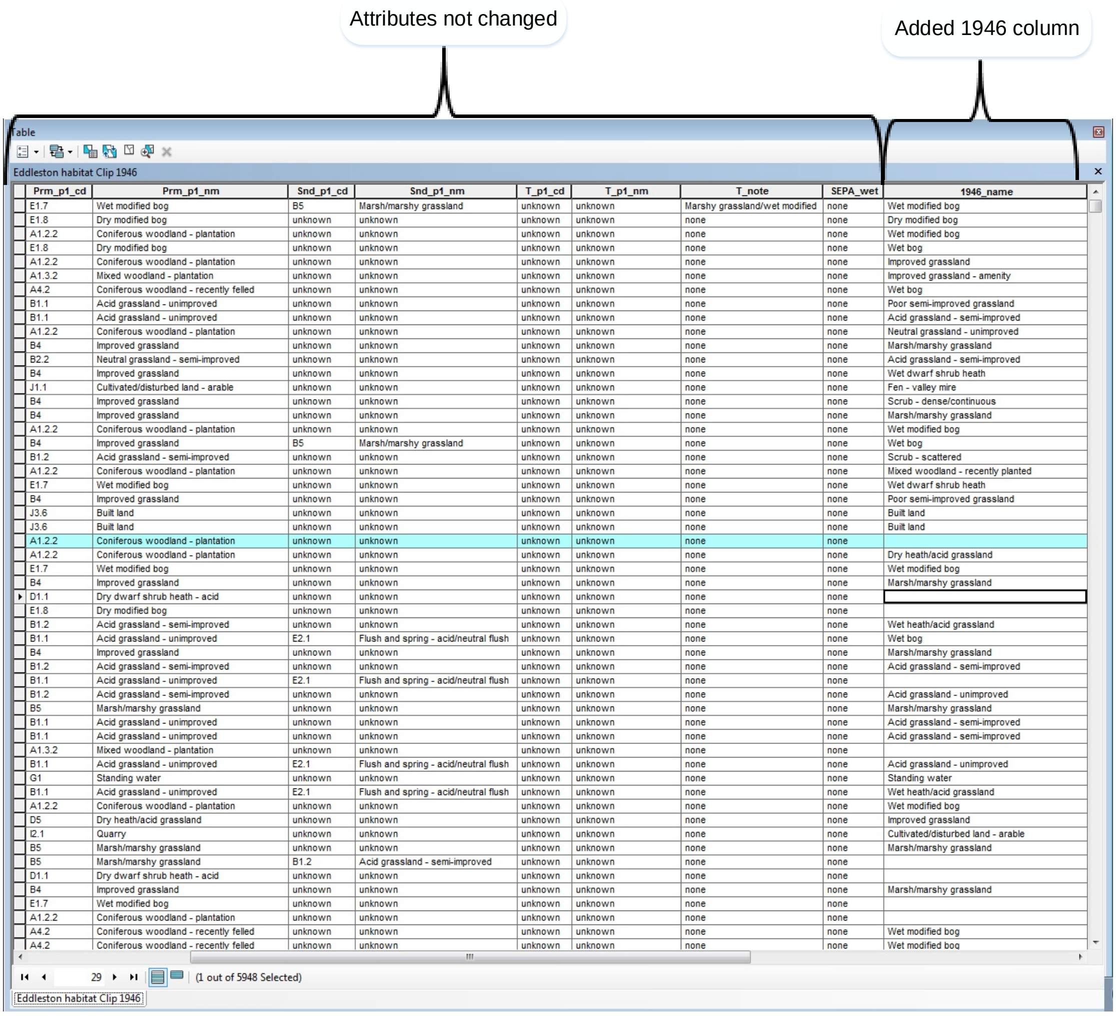


Figure 3-26 Air photo interpretation flow diagram

Alongside the interpretation process shown in the diagram above, the 1946 column in the attribute table for the habitat map was also populated with the interpreted habitat classes. The other attributes for the current (2009) habitat map which included the 2009 Phase 1 primary habitat class name, alphanumeric code, secondary habitat class name and target notes were retained as illustrated in the figure below.



Prm_p1_cd	Prm_p1_nm	Snd_p1_cd	Snd_p1_nm	T_p1_cd	T_p1_nm	T_note	SEPA_wet	1946_name
E1.7	Wet modified bog	B5	Marsh/marshy grassland	unknown	unknown	Marshy grassland/wet modified	none	Wet modified bog
E1.8	Dry modified bog	unknown	unknown	unknown	unknown	none	none	Dry modified bog
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Wet modified bog
E1.8	Dry modified bog	unknown	unknown	unknown	unknown	none	none	Wet bog
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Improved grassland
A1.3.2	Mixed woodland - plantation	unknown	unknown	unknown	unknown	none	none	Improved grassland - amenity
A4.2	Coniferous woodland - recently felled	unknown	unknown	unknown	unknown	none	none	Wet bog
B1.1	Acid grassland - unimproved	unknown	unknown	unknown	unknown	none	none	Poor semi-improved grassland
B1.1	Acid grassland - unimproved	unknown	unknown	unknown	unknown	none	none	Acid grassland - semi-improved
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Neutral grassland - unimproved
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
B2.2	Neutral grassland - semi-improved	unknown	unknown	unknown	unknown	none	none	Acid grassland - semi-improved
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Wet dwarf shrub heath
J1.1	Cultivated/disturbed land - arable	unknown	unknown	unknown	unknown	none	none	Fen - valley mire
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Scrub - dense/continuous
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Wet modified bog
B4	Improved grassland	B5	Marsh/marshy grassland	unknown	unknown	none	none	Wet bog
B1.2	Acid grassland - semi-improved	unknown	unknown	unknown	unknown	none	none	Scrub - scattered
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Mixed woodland - recently planted
E1.7	Wet modified bog	unknown	unknown	unknown	unknown	none	none	Wet dwarf shrub heath
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Poor semi-improved grassland
J3.6	Built land	unknown	unknown	unknown	unknown	none	none	Built land
J3.6	Built land	unknown	unknown	unknown	unknown	none	none	Built land
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Dry heath/acid grassland
E1.7	Wet modified bog	unknown	unknown	unknown	unknown	none	none	Wet modified bog
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
D1.1	Dry dwarf shrub heath - acid	unknown	unknown	unknown	unknown	none	none	
E1.8	Dry modified bog	unknown	unknown	unknown	unknown	none	none	
B1.2	Acid grassland - semi-improved	unknown	unknown	unknown	unknown	none	none	Wet heath/acid grassland
B1.1	Acid grassland - unimproved	E2.1	Flush and spring - acid/neutral flush	unknown	unknown	none	none	Wet bog
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
B1.2	Acid grassland - semi-improved	unknown	unknown	unknown	unknown	none	none	Acid grassland - semi-improved
B1.1	Acid grassland - unimproved	E2.1	Flush and spring - acid/neutral flush	unknown	unknown	none	none	
B1.2	Acid grassland - semi-improved	unknown	unknown	unknown	unknown	none	none	Acid grassland - unimproved
B5	Marsh/marshy grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
B1.1	Acid grassland - unimproved	unknown	unknown	unknown	unknown	none	none	Acid grassland - semi-improved
B1.1	Acid grassland - unimproved	unknown	unknown	unknown	unknown	none	none	Acid grassland - semi-improved
A1.3.2	Mixed woodland - plantation	unknown	unknown	unknown	unknown	none	none	
B1.1	Acid grassland - unimproved	E2.1	Flush and spring - acid/neutral flush	unknown	unknown	none	none	Acid grassland - unimproved
G1	Standing water	unknown	unknown	unknown	unknown	none	none	Standing water
B1.1	Acid grassland - unimproved	E2.1	Flush and spring - acid/neutral flush	unknown	unknown	none	none	Wet heath/acid grassland
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	Wet modified bog
D5	Dry heath/acid grassland	unknown	unknown	unknown	unknown	none	none	Improved grassland
I2.1	Quarry	unknown	unknown	unknown	unknown	none	none	Cultivated/disturbed land - arable
B5	Marsh/marshy grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
B5	Marsh/marshy grassland	B1.2	Acid grassland - semi-improved	unknown	unknown	none	none	
D1.1	Dry dwarf shrub heath - acid	unknown	unknown	unknown	unknown	none	none	
B4	Improved grassland	unknown	unknown	unknown	unknown	none	none	Marsh/marshy grassland
E1.7	Wet modified bog	unknown	unknown	unknown	unknown	none	none	
A1.2.2	Coniferous woodland - plantation	unknown	unknown	unknown	unknown	none	none	
A4.2	Coniferous woodland - recently felled	unknown	unknown	unknown	unknown	none	none	Wet modified bog
A4.2	Coniferous woodland - recently felled	unknown	unknown	unknown	unknown	none	none	Wet modified bog

Figure 3-27: Extract of the attribute table

The air photo interpretation steps illustrated in Figure 3-26 were consistently and repeatedly done until land parcels covered by air photos in both catchments were interpreted. The completion of the interpretation and editing process in ArcGIS yielded the 1946 habitat map for each catchment which were then used to map historic ecosystem services (stage 3) of data collection and processing.

Step 5: Print out the final 1946 habitat maps

The final step was to print out the final habitat maps for both catchments. This was done using the symbology function in ArcGIS which allows for selecting colour codes representing different habitat types. The legend, scale bar and north arrow were also added to the final printed maps. The generated and contemporary habitat maps for the study catchments are presented in Appendix 10a and 10b.

3.5 Stage 3: Ecosystem services mapping

This section first describes how mapping ecosystem services in current land cover/habitat proxy based studies is done. This is followed by a detailed description of how this study mapped ecosystem services including the ecosystem service mapping approach used.

3.5.1 Current practice in land cover based mapping of ecosystem services

As discussed in the literature review chapter, the common approach in current practice of mapping ecosystem services is the use of proxies to convert remotely sensed data such as land cover or habitat maps into ecosystem service maps (Science for Environment Policy, 2015). This is done at three tier levels depending on the amount of data sources used and level of detail presented. In the first tier, the land cover map is used as the only data source to map ecosystem services as illustrated in the figure below. For more detail and accuracy some studies go a step further into the second tier in which land cover maps are integrated with other data such as soil maps or primary data. The third tier adds another level of detail by including process based models which account for underlying processes (physical and biological) that affect ES supply. It incorporates numerous types of data including primary data and models such as the study done by Maes et al. (2012a), which mapped water purification services based on a model for nitrogen assessments. Figure 3-28 is an illustration of the basic (tier 1) steps in land cover/remotely sensed data based ecosystem services mapping approaches.

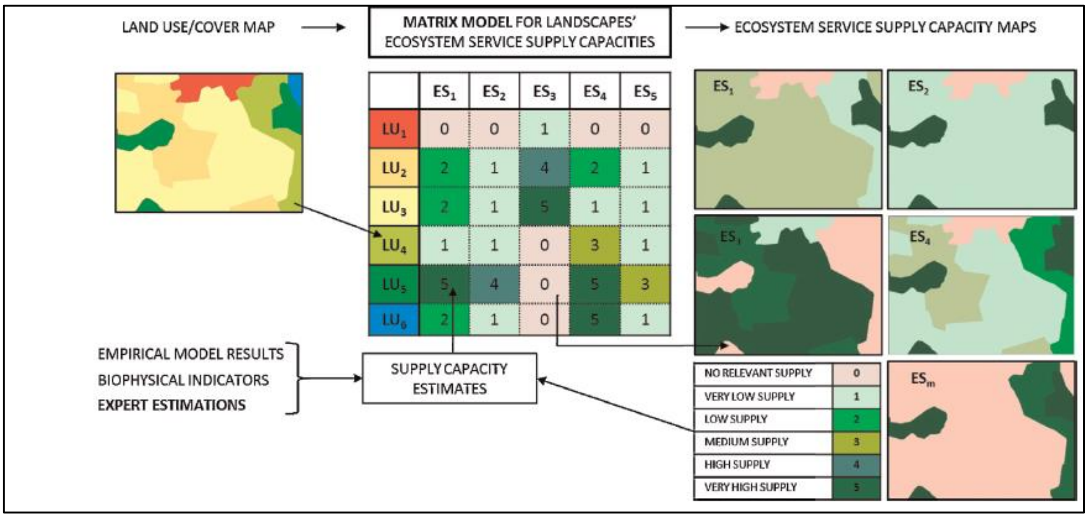


Figure 3-28: Basic steps in land cover based ecosystem services mapping approaches
Source: Jacobs et al. (2015)

As shown in figure 3-28, in the first step a land cover map is acquired which could, for example be the satellite derived CORINE land cover map. The second step involves formulating a matrix model/look up table by assessing the capacity of different land cover types to supply selected ecosystem services. Such an assessment is done qualitatively on a relative scale indicating ecosystem service supply capacity levels of different land cover types. For example, in the figure above, relative scale values ranging from 0 to 5 were used corresponding with the following qualitative explanations:

0 = shows areas with no relevant capacity to supply a particular ecosystem service,
1 = shows areas with very low relevant capacity to supply a particular ecosystem service,
2 = shows areas with low relevant capacity to supply a particular ecosystem service,
3 = shows areas with medium relevant capacity to supply a particular ecosystem service,
4 = shows areas with high relevant capacity to supply a particular ecosystem service,
5 = shows areas with a very high relevant capacity to supply a particular ecosystem service.

Relative scale value ranges differ with studies although the qualitative explanations are similar to those listed above. For example Frank et al. (2012) used relative scale value ranges from 0 (indicating areas with low ecosystem service supply capacity) to 100 (which indicated areas with very high ecosystem service supply capacity) while Haines-Young et al. (2012) used binary links of 0 (showing areas with a neutral role in ES delivery) and 1 (showing areas with a supportive role in ES delivery) and Vihervaara et al. (2010) used 0 (low), 1- medium and 2 for high capacity. The 0 to 5 scale shown in the figure above has typically been used by Burkhard et al. (2009), Burkhard et al. (2012) and Nedkov and Burkhard (2012).

These relative scoring systems are informed by expert estimations and scientific knowledge (literature review) on current understanding on ecosystem processes and the capacity of different land cover types to supply a particular ecosystem service (Jacobs et al., 2015, Haines-Young et al., 2012, Burkhard et al., 2009, Burkhard et al., 2012). All the example relative score value ranges given above can be understood as hypotheses and models that link different land cover types with ecosystem service supply capacities.

The last step involves the use of GIS to map the spatial location of ecosystem service supply areas. To do this, the relative score values from the look up table/matrix model are joined with the attribute table of the land cover map and then raster maths algorithms are used to produce ecosystem service maps. Different studies use different algorithms based on the relative scale value ranges used. Also in GIS, the land cover map can be integrated with other data layers such as soil maps for further detail and accuracy, thus moving into the second ecosystem services mapping tier.

3.5.2 SENCE ecosystem services mapping approach

The Spatial Evidence for Natural Capital Evaluation (SENCE) ecosystem services mapping method used in this study, is grounded on a mapping approach similar to that discussed above (Medcalf et al., 2014). It uses elements of both tier 1 and tier 2 with slight variations as discussed below: Firstly, SENCE uses habitat maps derived from the UK Phase 1 habitat classification system as the main underlying data source. Whereas this study might have used CORINE land cover maps, CORINE data is noted to be limiting due to its coarse spatial resolution if working at a local or regional scale (Burkhard et al., 2009). Secondly, SENCE has a matrix model/look up table informed by expert estimations and scientific knowledge and understanding (literature reviews). However, the look up table scores are adjusted to suit the study areas where ecosystem services are being mapped. For example, in mapping ecosystem services in the Scottish borders, where the study catchments for this study are located, the relative ES scores were presented to a local expert group consisting of habitat, soil and environmental experts who deliberated on these and adjusted them to suit the habitat types and land uses in the Scottish Borders (Medcalf et al., 2014). In this way local knowledge and conditions are taken into account in the SENCE methodology.

Thirdly, the SENCE method uses a five class relative scale of very low, low, medium, high and very high. In addition to the use of look up tables and depending on data availability, the SENCE method also integrates habitat maps with other existing spatial data layers in GIS e.g. soil map, topography, geology to capture the influence of such factors in ecosystem service delivery and this places it within the second ES mapping tier level. However, in cases where data are unavailable or limited, the habitat map is used as the only direct proxy for that particular ecosystem service (tier 1 level).

In SENCE the attribute table of the polygon habitat map is firstly joined with the look up table of relative values in GIS and then raster maths algorithms are used to map the selected ecosystem services. The multiple datasets are then blended/overlaid within the GIS to produce probable spatial representation of where in the landscape the selected ecosystem services can potentially be supplied. To develop individual ecosystem service maps, each of the data sets are standardized to produce a common algorithm to facilitate the overlaying process within the GIS environment. In such a standardisation process, the data layers to be combined are reduced to 0 or 1 raster maths matrices. To produce the final ecosystem service maps, the selected suite of standardized data sets are overlaid with the habitat maps to produce the corresponding extent maps for each selected ecosystem service (see Section 3.5.5). The final outputs from this method are ecosystem service supply maps, showing the relative importance of land parcels for ecosystem service supply, (as illustrated in Figure 3-28) for each of the selected ecosystem services. The figure (Figure 3-29) below outlines the SENCE ecosystem services mapping approach.

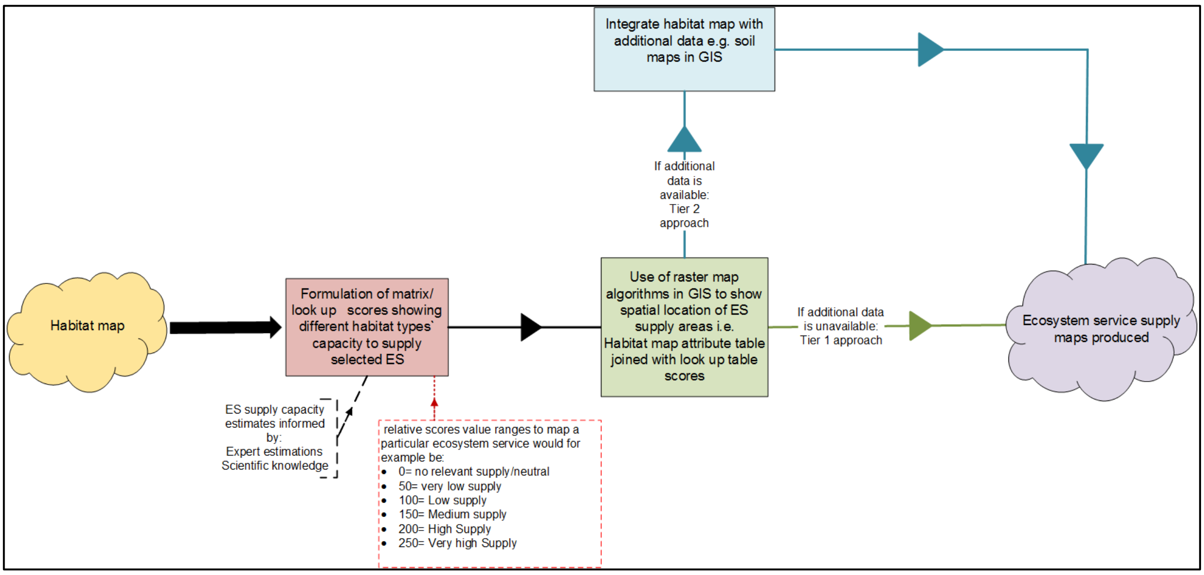


Figure 3-29: SENCE ecosystem services mapping approach
Adapted from Jacobs et al., (2015)

This study adopted the SENCE ES mapping method because (a) it better incorporates local variations, knowledge and locally available data for the study catchments, (b) it is applicable at the catchment level scale at which this study is done, (c) perhaps more fundamentally, this method is in line with current practice in land cover based ecosystem services mapping approaches, (d) since this is a comparative study, there is the need to use the same method used to map the current ecosystem services in the study catchments so as to minimise inaccuracies and errors arising from using a different ES mapping tool, and (e) the SENCE method has been chosen as a reliable ES mapping approach by conservation agencies in the UK such as JNCC, SNH and local authorities such as the Countryside Council for Wales (CCW; now Natural Resources Wales) and the Scottish Borders Council. Sections that follow describe how the SENCE method was applied to map historic ES in the study catchments.

3.5.3 The choice of historic ecosystem services to be mapped

The LUS pilot project in the Scottish Borders is a Scottish Government initiative to explore how the National land use strategy on mitigating the potential impacts of climate change could be implemented. The Scottish Government chose two pilot projects at a regional scale, Aberdeenshire and Scottish Borders, in which the responsible local authority in collaboration with partner organisations tested the potential implementation of the LUS through an Ecosystem Approach. In the Scottish Borders, the first step in implementing the pilot project was the baseline mapping of 17 of the most important ES provided by the current pattern of land use. The baseline mapping of the ES was done using the SENCE method (Spray, 2014). Six smaller sub catchments were selected within the Scottish Borders in which detailed ES maps were produced. These included the Ale and Eddleston catchments which this study also focusses on. These detailed ES maps were used for comparison with the historic ES produced in this study.

Ten out of the possible 17 ecosystem services were selected for mapping in each catchment (refer to the table 3-2). These were ecosystem services that were identified as important and a priority in the Scottish Borders by local stakeholders during the LUS pilot project stakeholder consultation process (Spray, 2014). This study then sought to analyse whether the prioritised ecosystem services have changed between 1946 and 2009, with a further aim of understanding the drivers of these changes and the implications of such changes. Some of the selected ecosystem services have of late gained national and global significance and policy relevance e.g. carbon storage. In addition, data was available from earlier times such that it would be feasible to map from the historic habitat maps and to focus analysis on change over time.

Table 3-2: Ecosystem services selected for mapping in this study

UK NEA ES category	ES mapped	Why selected
Provisioning	Food: Agricultural crops	Agriculture is one of the most dominant land uses in the Scottish Borders (over 80% of land is under agriculture) and this reflects its importance in this area. It was selected to assess how this major ES has changed and responded to both policy and non-policy drivers of change.
	Food: Agricultural Livestock	
	Trees: Timber resource	There have been substantial woodland plantations and it was of interest to map the areas where such plantations have been introduced and analyse the ES traded off when woodland plantations were introduced.
Regulating	Climate regulation: Soil carbon storage	This ES is of national and global relevance as there is marked interest in carbon storage prompted by climate change issues. Analysing how these have changed over time would inform the ongoing climate change adaptation measures.
	Climate regulation: Vegetation carbon storage	
	Detoxification and purification: Water quality	Water in streams, rivers, lochs and reservoirs are characteristic to this area and there are also issues related to diffuse pollution and fragmentation of small wetlands. It was of interest to map and analyse the changes to these overtime.
	Pollination	This is one of the important supporting ES. There is increased interest in pollinator habitats as these greatly contribute to agricultural production and analysing whether these have changed overtime would provide valuable pointers.
	Soil quality: Land erosion risk	Given the dominance of agricultural production in these areas, it’s important to highlight high soil erosion risk areas and to analyse how they compare between the two time periods and

UK NEA ES category	ES mapped	Why selected
		how this could inform soil erosion control measures.
	Water regulation: Water quantity	This is of policy relevance especially in natural flood management and it would be of benefit to map such areas with high potential and help to identify potential areas for flood control measures which were in place in 1946.
Supporting	Wild species diversity: biodiverse habitats	There are many biodiversity protected sites in the study areas. Mapping such areas would help in identifying areas important for biodiversity and nature conservation. This would also be of value in habitat restoration measures as this would show the size, connectivity of habitats in 1946 and could inform the current measures to restore the habitat networks in order to conserve biodiversity

In the LUS pilot project, 17 ES were identified and mapped as important and prioritised in the Scottish Borders (Spray, 2014). Seven of these ecosystem services (refer to table 3-3) were not mapped in this study. These ES were excluded because data was not available to map them for past time periods. For example, cultural ecosystem services could not be mapped in retrospect because there wasn't enough data applicable to 1946 that could have been used to map these using the SENCE methodology and these are also best understood through capturing stakeholder perceptions. In addition, ecosystem services like energy (renewable) were not relevant at that time. The table below gives a list of the ES that were excluded from historic ES mapping.

Table 3-3: Ecosystem services excluded in historic ecosystem service mapping

ES excluded	Reasons for exclusion
Energy (renewable)	-Some ES were not regarded as relevant in the past e.g. wind energy. -Comparable data not available for 1946 or would have been difficult to access for these ecosystem services. Examples of such data include: SEPA flood risk data layers and designations
Flood risk	
Landscapes	
Local places	
Biodiversity and resilience networks	

Historic and archaeological significance	(national scenic areas, natural heritage zones, parks, visitor attractions, riding routes, listed buildings, scheduled monuments etc.)
Recreation (non-motorised)	
Recreation (sporting)	

3.5.4 The ecosystem services typology adopted:

Maes et al. (2013) advise that ES mapping attempts should specify the ES typology used given the different typologies used to define ES (as discussed in the literature review chapter). This study adopted the UK NEA typology (which essentially follows the MEA typology). The UK NEA nomenclature was selected chiefly because this study was conducted in the UK and hence a nationally used typology was adopted. Also, for consistency, the UK NEA typology was to produce the current ES maps for the study catchments.

3.5.5 Historic ecosystem services mapping: the procedure

The 1946 habitat maps i.e. the output from the habitat mapping stage (stage 2) were the main data sets used to map historic ES. The historic habitat maps were combined with different datasets in System for Automated Geoscientific Analyses Geographic Information System (SAGA GIS) to map each of the selected ES. SAGA GIS is an open source software (SAGA, 2015) which offered a cheap and effective way of implementing spatial algorithms such as those used in the SENCE methodology.

1 • In MS excel, ES scores/relative values for each of the ES to be mapped to the 1946 habitat types are assigned using supplied look up tables

2 • Import matrix tables for each subcatchment from MS excel into ArcGIS and join them with the respective 1946 habitat layer attribute table

3 • Import and convert the joined 1946 habitat layer (vector format) into a ES raster layer in SAGA GIS

4 • Upload and clip the Scottish Borders underlying base maps (aggregated datasets) to sub catchment boundaries in SAGA GIS

5 • Superimpose the raster layer (1946 layer for each ES) on corresponding underlying basemap/datasets to get the final output map for each ES

6 • Export the final ES map into ArcGIS, edit layout and print the final ES map

7 • Repeat steps 3 to 6 to map each ES at a time for each sub catchment to get a total of 10 ES maps per catchment and 20 ES maps in total.

Figure 3-30: Overview of the historic ES mapping procedure

Step 1: Assign ES scores/relative values to the 1946 habitat classes using supplied look up tables

Firstly, all the 1946 mapped habitat types for each of the catchments were listed in Microsoft excel. Using the look up table provided by Environment Systems Ltd, each of these habitat types were assigned a relative value/score for each of the ES to be mapped. Assigning of these values was based on the potential of a particular habitat type to provide that particular ES. This was based on the assumption that the value of an ecosystem service is constant[10] for a particular habitat type/class. For example, arable agriculture fields scored high on crop production ES, while built land was coded as inapplicable as it does not provide this ES. Equally bogs and heathland scored high on biodiversity ES, while improved grassland scored low. This resulted in the formulation of matrix tables (refer to appendix 7) and these were saved as csv files compatible with ArcGIS file formats.

Step 2: Import matrix tables from MS excel into ArcGIS and join with 1946 habitat layers attribute table

The matrix tables (saved as csv files) were imported into ArcGIS. ArcGIS was used here because the join function in SAGA GIS proved to be a challenge and ArcGIS offered an easier alternative. So, using the join table function in ArcGIS the matrix table for each catchment was joined with the respective attribute table for the 1946 habitat layers. The joined attribute table for the 1946 habitat layers (with the ES scores) were saved with a different name to differentiate them from the original 1946 habitat layers (without the ES scores) and used in the successive steps below.

Step 3: Import and convert 1946 habitat layer (vector format) into series of ES raster layers in SAGA GIS

The 1946 habitat layers (with assigned ES scores) formed in the previous step, were then imported into SAGA GIS. Using the gridding function in SAGA GIS, the 1946 habitat maps for each catchment were converted into a series of ES layers for all the ES selected for mapping. The gridding function allows for the conversion of vector data into raster format using the ES scores assigned in step 1 to provide a continuous representation of the distribution of each of the ES in the study areas. The conversion of these habitat layers

[10] Underlying assumption in ecosystem services mapping.

into ES raster layers was done separately for each ES as these had different scores used to map that particular ES. Thus the drop down menu in the SAGA GIS gridding function gave an option to select the ES to map. This is illustrated in the screen shot below.

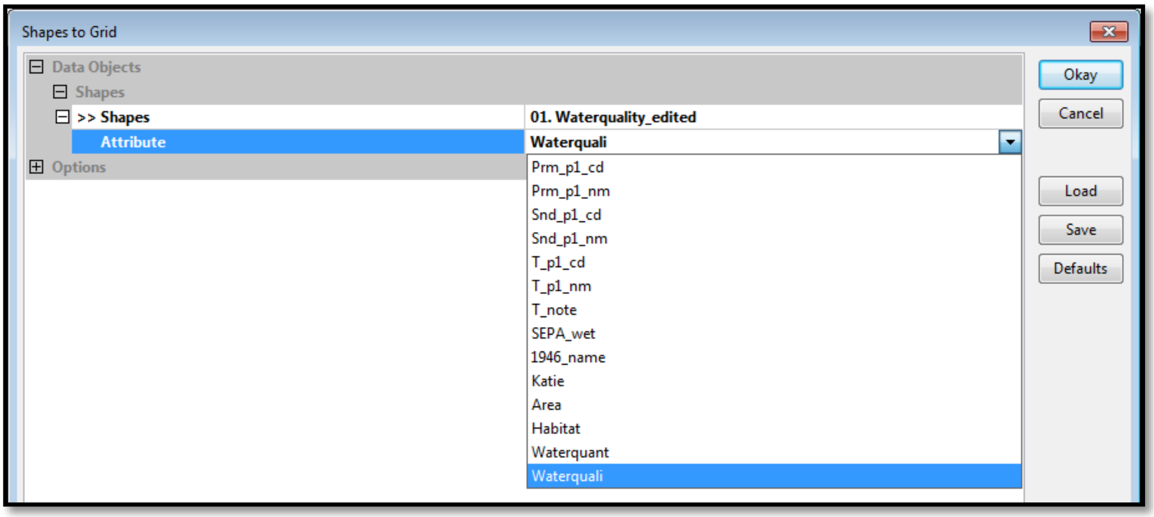


Figure 3-31: Using the drop down menu from the gridding function in SAGA GIS

As shown in figure 3-31, the highlighted water quality ES was selected so that the vector version (shape) could be converted into a raster water quality ES layer i.e. highlighting areas with a high potential to provide this ES based on the assigned ES scores (pixels have different values). Figure[11] 3-32 gives a further illustration of the output from converting the vector habitat layer into the raster water quality ES layer. The pixel size of these maps was 10 m owing to the high resolution and scale of the original data source used in this study i.e. aerial photography (1: 10 000).

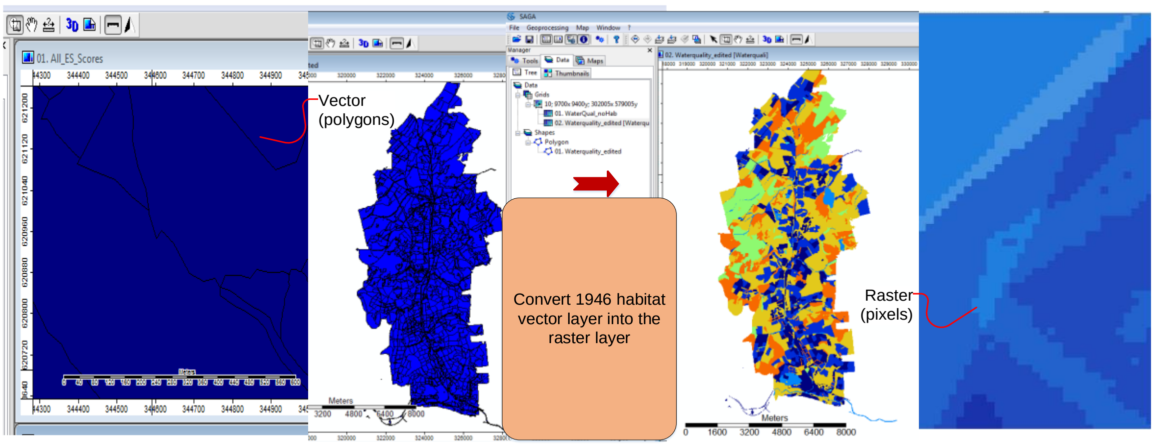


Figure 3-32: Conversion of the habitat vector layer into the water quality ES raster layer

[11] The Eddleston is being used as an example in this case, the same procedure was done for the Ale catchment

As illustrated in the figure above, the habitat vector layer was imported as a shape file (refer to the left pane in the figure) and this was converted into the water quality ES raster layer (refer to the right pane in the figure). The different colours in the raster layer (on the right), indicate the potential of the habitat types in the Eddleston catchment to deliver the water quality ES. These colours also correspond to the scores that were assigned to these habitat classes in step 1.

Step 4: Upload and clip the Scottish Borders underlying base maps (aggregated datasets) to catchment boundaries in SAGA GIS

The raster file which contained the aggregated datasets for each ES was also loaded in SAGA GIS and this formed the underlying base map on which the 1946 ES raster layers from the previous step were superimposed. Since different datasets were used to map each ES (refer to appendix 6), this implies that the underlying base maps differed with type of ES being mapped - for example the underlying base map for water quantity regulation ES was a blend of the National soil inventory for Scotland, the BGS superficial and bedrock data, and the DTM. This contained information about the slope, geology, soil and drainage which then had to be combined with the 1946 water quantity ES raster layer to indicate the potential of the various habitat types in each of the catchments to deliver this ES. Figure 3-33 shows the aggregated datasets (other spatial data) used to map water quality ES (excluding the habitat layer). These aggregated datasets[12] were produced during the ecosystem services mapping phase of the LUS pilot project and they cover the whole of the Scottish Borders. In this study, these had to be clipped (i.e. using the clip grid function in SAGA) to the study catchment boundaries as illustrated in figure 3-30.

[12] The underlying factors of soil, geology, slope are assumed to have remained the same (constant) between 1946 and 2009

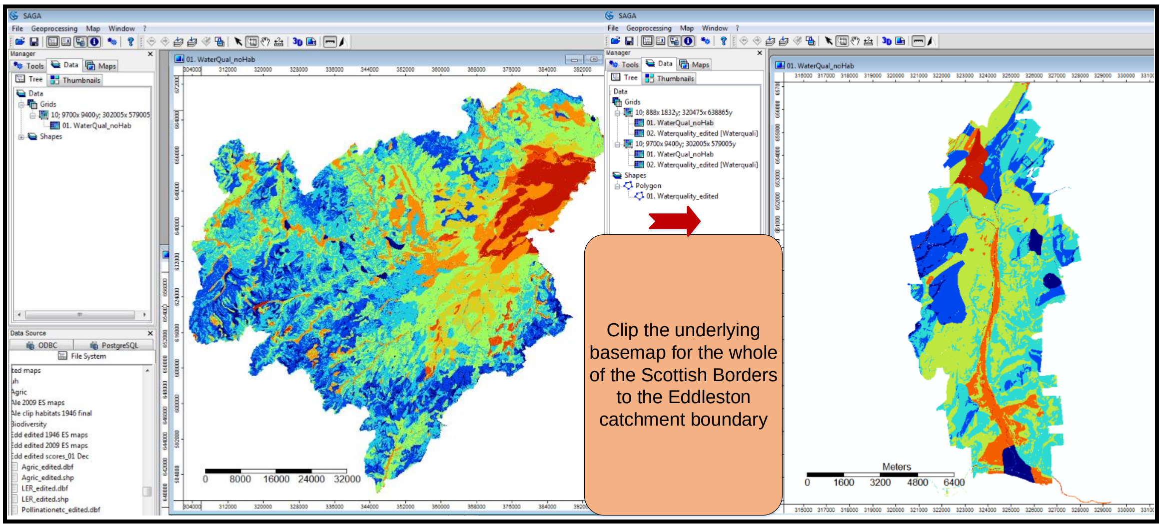


Figure 3-33: Underlying aggregated spatial data clipped to the catchment boundary

The figure shows the water quality ES aggregated spatial data for the whole of the Scottish Borders (on the left), which was clipped to the Eddleston catchment boundary (on the right). The underlying spatial data corresponding to each of the ES mapped were clipped to catchment level.

Step 5: Superimpose the raster layers (1946 raster layer for each ES) on corresponding underlying aggregated spatial data to get the final output maps for each ES.

In order to construct the final ES map, the 1946 raster layer (from step 3) was combined with the corresponding underlying aggregated datasets (from step 4), using the grid calculator (calculus) function in SAGA GIS (see figure 3-34).

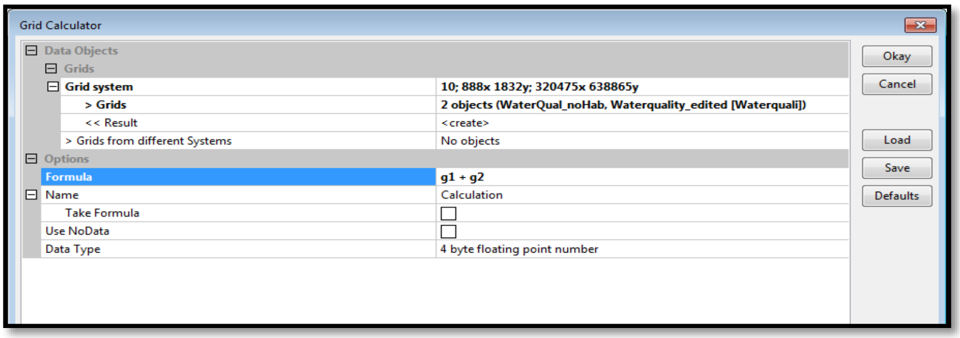


Figure 3-34**:** Grid calculator function in SAGA GIS

The Grid calculator function, gives an option to enter the formula to be used and also to select the raster layers to be combined together using the defined grid system (Refer to figure above). Figure 3-35 below is an illustration of the integration of the layers to generate the final water quality ES map.

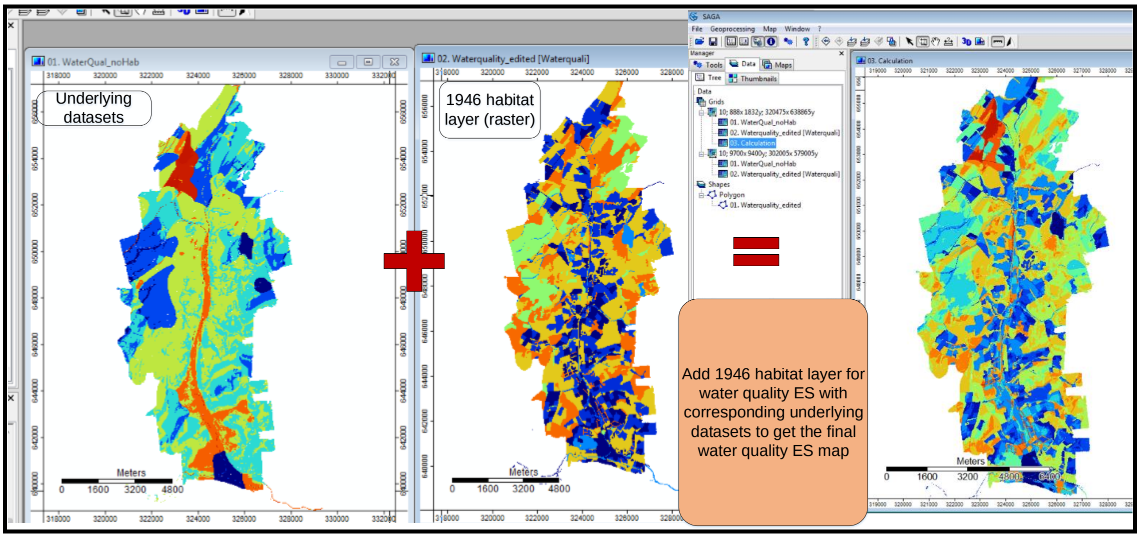


Figure 3-35**:** Raster layers added together to get the final 1946 water quality ES map

The raster layers from step 3 (the middle one) and step 4 (on the left) were superimposed on each other/blended to produce the final 1946 water quality ES map (on the right) as demonstrated in the figure above. Combining these layers assesses how the influencing factors of soil/geology, landform, management and habitat interact to deliver an ecosystem service.

However, this step was not done for all the 1946 ES raster layers. This is because, for some ES accompanying underlying datasets were purposely excluded for historic mapping as the data sets used to formulate these aggregated spatial data was not relevant/applicable for 1946. For example, in mapping the current crop production ecosystem services, the data sets used included the allotments and management IACS permanent data and the phase 1 habitat layer. In this case the allotments and management IACS data were not applicable for 1946 and so the 1946 habitat map was used as a direct proxy for this ES. This was also the case for livestock and timber production. Thus the 1946 raster ES layers (from step 3) were exported as the final ES maps for these. Table 3-4 gives a list of the ES maps that were produced after combining the 1946 habitat layers with the corresponding underlying spatial data sets (tier 2 approach as illustrated in Figure 3-29, section 3.5.2) as well as the list of the ES in which the habitat map was used as a direct proxy for that ES (tier 1 approach as illustrated in Figure 3-29, section 3.5.2).

Table 3-4: List of ecosystem services that were mapped using the SENCE tier 1 and tier 2 levels

ES maps produced by combining habitat maps with other spatial data (Tier 2)	ES maps in which habitat maps were used as a direct proxy (Tier 1)

Water quality	Crop production
Water quantity	Livestock production
Soil carbon storage	Timber production
Biodiversity	
Land erosion risk	
Vegetation carbon storage	
Pollination	

Step 6: Export the final ES maps into ArcGIS to edit the map layout

The final step was to export the final ES maps (.asc file format) into ArcGIS. In total 10 ES maps per catchment were produced to match the initially selected 10 ES (20 in total) (presented in the results chapter). These were uploaded in ArcGIS so that the preferred colour legends could be selected (see figure 3-36). The gradient of light to dark colour shades were selected to correspond to low and high levels of ES delivery. Each ES map for each catchment was edited to include the map layout (North arrow and scale bars) and printed.

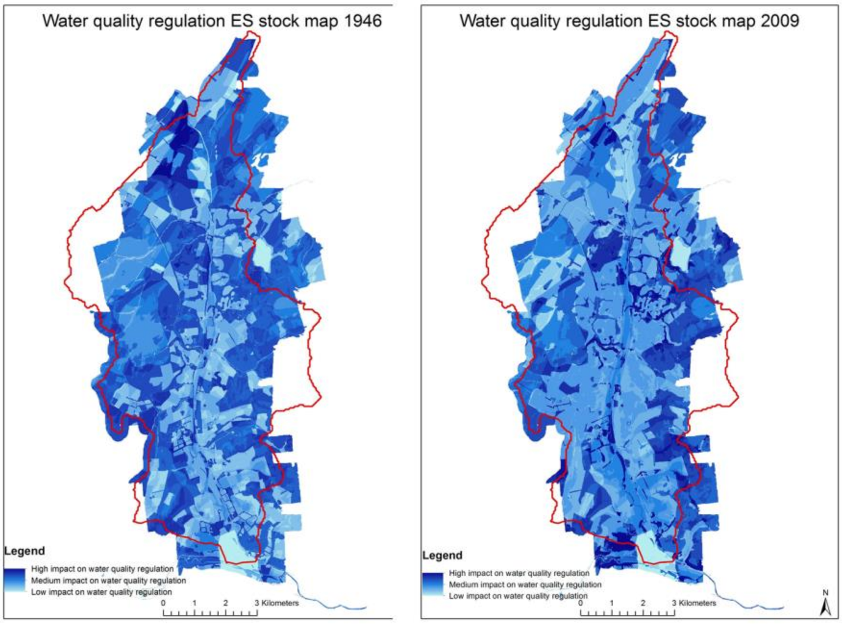


Figure 3-36: 1946 and 2009 water quality final ES maps printed out

3.6 Uncertainties and limitations of the study

Many authors (e.g. Cherrill and McClean 1999a; Lechner et al. 2012; The Macaulay Land Use Research Institute 1993; Congalton and Green 2009) acknowledge that all databases, either spatial or otherwise, are likely to contain a certain degree of uncertainty and hence it is important that the scale and nature of this is known especially to the end users. Sources of uncertainty and errors could be attributed to input data used, the rules and methods of mapping both habitats and ecosystem services (Lucas et al., 2011). Uncertainties associated with each of the stages are discussed below and how this could have influenced the following subsequent stages. Measures implemented to minimise these errors and accuracy assessment procedures done are also included for each of the stages.

3.6.1 Stage 1: Processing of historic aerial photographs

Radiometric and geometric/positional errors are basic aerial photographic errors (Morgan et al., 2010, Verhoeven et al., 2012b, Lunetta and Congalton, 1991). Radiometric errors impact on the colour tone of the aerial photography (air photo quality). Factors that contribute to this type of error include: the vantage point, types of filters used during air photo capture, and the condition and calibration of the camera used. Season and time of image capture also impact on the air photo colour tone. Geometric errors alter the perceived location and size of features on a photo. Such alterations are influenced by the stability of the platform during image acquisition and the type of equipment used to capture the air photos e.g. camera lens distortion.

Since this study relied on historic aerial photographs, some of these had radiometric errors. To minimise the use of such poor quality air photos, photos with acceptable levels of clarity and colour tones were opted for, as these were suitable for habitat mapping. One of the criteria used during the air photo searching and gathering process included considering high resolution, clear, large scale and good quality air photos. However, since the most important criteria in the selection of the historic photos was the time period when they were taken, some of the photos selected though of high resolution and large scale had varying levels of illumination. Such air photos were selected in situations where they were taken in the 1940s and no better alternatives from the same period were available.

Photos that were identified as outliers/stray photos in the photo alignment step (Figure 3-15) were a result of geometric errors as it is presumed that they were captured at a tilted look angle. The process of editing and removing these photos, as a way of minimising this type of error, led to a cut back on the orthophoto areal extent and hence slightly reduced the size of the catchment areas reconstructed. In addition, a small part of the Ale catchment was also omitted as there were no photos taken in the 1940s that covered these areas. However, these alterations were of less concern as over 90% of catchment areas were reconstructed.

Orthorectification addresses geometric errors (Morgan et al., 2010). In this study, the orthorectification procedure of selecting, recording and assigning the GCPs was manually done in ArcGIS and Photoscan. Since this was done manually, errors could still have emanated in this process, especially given the difficulty of identifying and matching GCPs on historic air photos caused by land cover and feature position changes. Error estimation in Photoscan yielded a total horizontal Root-Mean-Square Error (RMSE) of 4.8 m for the Eddleston and 4.5 m for the Ale. This was the reprojection error for each GCP calculated over all photos where the GCP was visible (Agisoft, 2013). These RMSE values are consistent with RMSE values realised in studies that have done similar orthorectification procedures on historic aerial photographs such as Hughes et al. (2006) and Ouedraogo et al. (2014). For example, Hughes et al. (2006) realised RMSE accuracy estimations of ± 5m, which was considered a high degree of accuracy.

The overall accuracy and suitability of the generated orthophotos was assessed by checking their positional accuracy. This was done by overlaying the orthophotos with current aerial photography in ArcGIS to check whether these layers were spatially aligned. This was crucial and arguably very important in this study as the ability to interpret and map habitats relied on how accurate these data layers corresponded. Figure 3-37 below illustrates the positional accuracy of the orthophotos generated in this study.

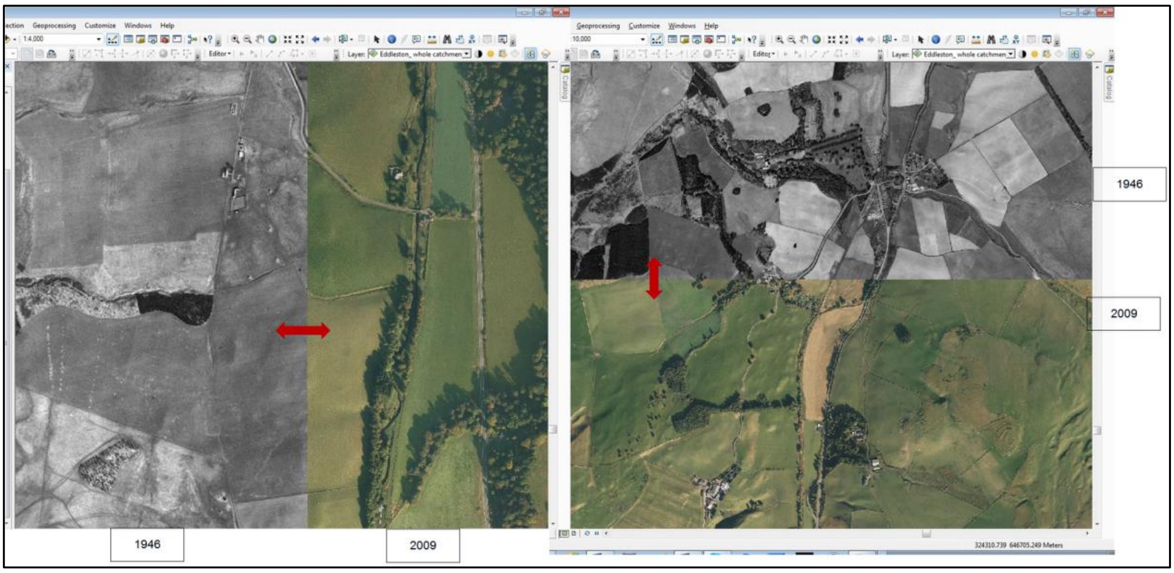


Figure 3-37: Positional accuracy of the generated orthophotos- alignment of features 1946 and 2009

Figure 3-37 shows that the orthophotos neatly overlaid on the current colour aerial photography as seen from corresponding and matching boundaries, roads and river streams. Based on this, the generated orthophotos were considered highly accurate, suitable for interpretation and reliably inform habitat mapping.

3.6.2 Stage 2: Air photo interpretation and habitat mapping

The observer error (also referred to as the attribute classification error or misclassification error) and the boundary error (also referred to as the positional error or spatial displacement of boundaries error) are the common basic type of errors occurring in habitat mapping (Cherrill and McClean, 1999a, Lechner et al., 2012, Joint Nature Conservation Committee, 2010). An observer error occurs when the interpreter assigns a parcel of land to an inappropriate habitat class while the boundary error results from misplacement of boundaries between parcels of different habitat types.

Encounters which led to observer errors during habitat mapping in this study were related to the difficulty of interpreting shades of gray in black and white air photos. In particular, it was difficult to consistently differentiate some habitat classes, especially in cases where they were distorted by photo quality (i.e. radiometric errors). At times, for example it was difficult to interpret whether the woodland type was broadleaved or coniferous owing to poor photo illumination. Other habitat classes such as roads and streams were overshadowed by trees and hence it was at times a challenge to clearly trace their boundaries.

Even though evidence of plough lines was used as one of the criteria for identifying cultivated/disturbed arable land, it was difficult to consistently differentiate between improved grassland and cultivated/disturbed land due to almost identical tonal and textural patterns on some air photos. These habitat classes could have also been influenced by the agricultural activities undertaken at that time or by the types of crops grown during that season (crop rotation). The confusion between improved grassland and cultivated/disturbed land is acknowledged to be a common challenge with air photos taken around May/June as this was also encountered in the LCS88 air photo interpretation survey (The Macaulay Land Use Research Institute, 1993). Also, in the uplands the criteria used to identify modified bogs was based on the evidence of drainage lines but in some cases this was confusing as some acid grasslands also had drainage lines.

Mapping of complex/transitional/mixed habitat mosaics posed another challenge. This was also compounded by the fact that the Phase 1 classification system excludes mosaics which constitute of more than two habitat types. Such complex mosaics were mainly found in the uplands, especially heath and acid grassland combinations or heath found in association with a modified bog or flush, and spring, heath and bogs. Similarly, the difference between dwarf shrub heath and heath was difficult to distinguish from the air photos.

In relation to the boundary error, placing boundaries between unenclosed habitat types/classes especially in the uplands was a challenge. Fuzzy lines were drawn to represent assumed boundaries and this could have contributed to the boundary errors in this study. Some habitat types e.g. grasslands were in a continuum such that it was difficult to define their boundaries and hence contributing to the boundary error. It is indeed accepted that it is difficult to draw artificial boundaries along gradients of continuously varying natural or semi-natural habitat/vegetation (Cherrill and McClean, 1995, Jarman et al., 2010).

Another limitation of the habitat mapping approach adopted was due to the fact that all features in the habitat maps were represented as polygons. The decision to present all features as polygons relied on the feature format of the current (2009) habitat map which was used as a guide to derive the 1946 habitat maps. This included habitat types such as rivers, roads and hedgerows that could have been represented as line features. As a result, tracing out the boundaries of such habitat classes might have under or overestimated their actual boundaries on the ground, contributing to boundary errors.

A major inherent challenge associated with habitat mapping is the fact that the landscape is a continuum rather than a series of discrete classes (Taylor et al., 2000). Because of this, subjectivity was also inevitable in this study as interpreters have to make judgements in order to map landscape elements as discrete classes. Also, some of the above discussed challenges and points of confusion encountered were partly related to the limitations of the Phase 1 habitat mapping method. For example, in the Phase 1 handbook manual the difficulty of distinguishing different grassland types is acknowledged, but the manual does not provide further guidance on how interpreters/surveyors can explicitly differentiate these. This means interpreters have to make personal judgments and in the process this introduces subjectivity, as was also observed in the studies done by Cherrill (1995) and Cherrill and McClean (1999b). It is recognised that the backdating approach adopted in this study and the use of current habitat maps in air photo interpretation will also have influenced perceptions, especially on unclear and confusing habitat types.

While some of the error sources and challenges were inevitable, quality control measures were introduced to minimise these. Such measures included consulting and verifying points of confusion with an ecologist experienced in air photo interpretation and familiar with the study catchments. The following section describes procedures undertaken to assess the accuracy of the generated habitat maps.

3.6.3 Accuracy assessment of air photo interpretation and habitat mapping

Assessing the accuracy of generated historic maps was not possible as there was no temporally equivalent reference data against which the historic maps could be compared. However, accuracy procedures were undertaken to verify the accuracy of the current habitat maps. To do this, a field visit and previous air photo interpretation data was used to compared with the habitat maps. These procedures were viewed as more of quality control than typical accuracy assessment measures. Previous interpretations, field surveys or other available thematic data are considered appropriate reference data (Foody, 2002, Morgan et al., 2010).

Following Congalton (2001), a basic error matrix (also referred to as the confusion or agreement matrix) was used to summarise observations from these verification procedures. Prior to these, a visual assessment of the generated habitat maps was undertaken to make sure that the maps looked correct. It involved inspecting the spatial orientation of maps by checking whether the position of roads, settlements and other typical features to the study catchments were correctly represented. Such an inspection showed that the habitat maps were correct. Visual inspection is a recognised basic accuracy assessment procedure (Olofsson et al., 2014, Congalton, 2001). However, Congalton (2001) cautions that doing it alone does not suffice in accuracy assessment.

In this regard, a field visit to the Ale catchment was undertaken while previous air photo interpretation land cover data (1988 Land Cover of Scotland air photo interpretation) was used for comparing the Eddleston catchment habitat maps. Below is a description of the procedures followed:

1) Field verification in the Ale catchment

The field visit was done in summer (13 August 2014) with the assistance of an experienced ecologist, familiar with the catchment. The middle section of the Ale catchment was selected as the strata for the verification exercise as it has typical habitat types found in both upper and lower sections of this catchment. The sampling scheme adopted was a combined approach (stratification and cluster sampling) in which the selected strata (habitat classes) within the middle catchment represented habitat classes typical to this catchment.

Verification of habitat clusters within this strata was determined by practical accessibility within the area i.e. ease of accessing walking paths and access roads. As a result, the sampling rate was high alongside roads and walking paths. A walking distance of about 10km was covered during the verification exercise (Figure 3-38) and a drive through parts of the catchment was used to verify habitat classes that had distinct boundaries e.g. fields and built up places. The approach of combining stratification and cluster sampling has been advocated for situations where the estimation of class specific accuracy such as habitat classes in this case, is of prime importance (see for instance: Stehman et al. (2008)). It is also recognised that factors like costs and physical access influence the choice of the sampling design (Foody, 2002, Congalton and Green, 2009).

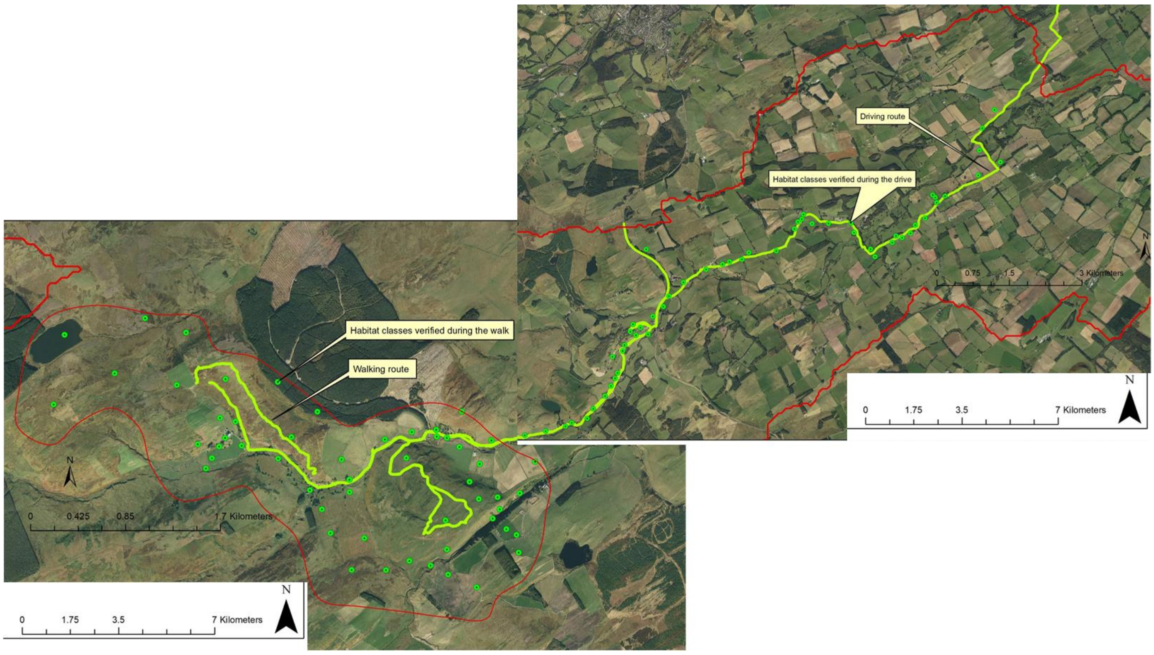


Figure 3-38: Field verification route in the Ale sub catchment

The left bottom picture shows the spatial location of the habitat classes that were verified during the walk in the uplands of the middle section of this catchment. The higher altitude gave vantage points in which a number of habitats could be verified and these were mainly the semi-natural habitat mosaics such as bracken, heath and acid grasslands. The picture on the right shows the driving route followed. This was the flat terrain in which habitat types close to the road could be verified and these were mostly habitats within distinctive boundaries such as arable fields and woodland plantations.

A touch field book computer was used to capture the verification data. Prior to the field visit, QGIS software in the touch field book computer was loaded with the main dataset layers used in the onscreen interpretation i.e. the Ale 1946 orthophoto, 1946 habitat map, 2009 aerial photography and the 2009 habitat map. A vector layer, capturing agreements and disagreements between 2009 habitat classes and field observations was populated during the visits.

An error matrix proposed by Congalton (2001), was used to summarise these observations as shown in Appendix 11a. In total, 116 polygons/habitats classes were verified and of these, 105 polygons lie on the leading diagonal while the remaining were off the diagonal. Numbers in the leading diagonal represented agreement between 2009 habitat classes and field observations (appendix 11a). This gave the overall accuracy estimation of approximately 91%. Off diagonal habitat classes indicated misclassification errors resulting from confusion in interpreting complex habitat mosaics, especially the semi-natural grassland mosaics such as marshy grassland and unimproved acid grassland.

2) Use of the LCS88 air photo interpretation as reference data for Eddleston

The land cover map for the whole of Scotland derived from the 1988 air photo interpretation done by the then Macaulay Research Institute was used to verify the Eddleston 2009 habitat map. This was another available land cover reference data derived from air photo interpretation, which this study could use to compare with the current habitat map. Comparison of this data was desktop based.

Stratified random sampling was used to select polygons to be compared. To select the strata, the Eddleston valley floor was selected as one stratum for validating the habitat classes typical to this part of the catchment, while the uplands on either sides of the valley were selected as another stratum representing typical habitat classes. The habitat maps were overlaid on the 1988 land cover data layer and the level of agreement and disagreement between the 2009 habitat classes and LCS88 habitat classes was captured following the same procedure done during the field visit in the Ale catchment.

The level of agreement and disagreement between the 2009 habitat classes and LCS88 habitat classes was summarised using an error matrix as shown in appendix 11b. In total, 83 polygons/habitats classes were verified and of these, 74 polygons lie on the leading diagonal while the remaining were off the diagonal. As explained above, numbers in the leading diagonal represented agreement between 2009 and 1988 interpretation (appendix 11b). This gave the overall accuracy estimation of approximately 88%. Similar to the Ale catchment, high agreement was observed for distinctive habitat types e.g. built up areas.

However, it was a challenge to align the 1988 (reference) polygon boundaries with those from this study owing to the scale with which these were produced. The 1988 land cover data was generated at a smaller scale as this covered the whole of Scotland while the habitat maps for this study were done at a larger scale and captured finer details than the reference data.

3.6.4 Stage 3: Historic ecosystem services mapping

The main type of error in ES mapping is the generalisation error (Eigenbrod et al., 2010b). This is argued to be an inherent source of error in ES mapping as it emanates from the underlying simplifying assumption in ES mapping (Eigenbrod et al., 2010a). This is that the ES delivery value is constant for the same habitat classes across the entire area being mapped. This value is also assumed to be the same in the area being mapped as in the studies from which the value was obtained.

This similar assumption was held in mapping ecosystem services in this study, as it was assumed that this value was constant between the two time periods (1946 and 2009). The look up tables used to assign ES scores in mapping the current ES were also used to assign ES scores for 1946 and these values were assumed to be constant. For example, the scrub habitat classes found in the Eddleston were assigned a similar constant value for the same habitat classes found in the Ale catchment. It was also assumed that the delivery of, for example, the crop and livestock production ES was constant in all fields labelled under these habitat classes yet different crop varieties give different yields. Thus adopting this assumption in this study also introduced the generalisation error.

This assumption was adopted in this study in order to maintain the same ES mapping approach (SENCE method) used to map the current ES in the catchments and eliminate possible discrepancies resulting from using a different approach since the intention of this thesis was to compare changes in ecosystem service delivery. This could be accepted as one of the limitations of the ES mapping method used and indeed a widely accepted one given that this assumption has been adopted in most ES mapping studies (Eigenbrod et al., 2010a) . Complexities of mapping ecological processes and functions are cited as the main reason underlying the adoption of this simplifying assumption in ES mapping. In reality, however, different habitats across space and time have varying biophysical characteristics, geographic variations among ES and hence varying ES delivery efficiency (Eigenbrod et al., 2010a).

Another limitation inherited from the habitat mapping stage was that habitats below the minimum mapping unit such as small streams and individual trees were in turn also not mapped into ES. Yet such habitats actually contribute to the delivery of important ES such as biodiversity, pollination etc.

Due to data limitations, some ecosystem services such as the cultural ones were not captured in the maps, yet they were identified as one of the important ES in these study catchments. Also linked to this, the 1946 habitat map was used as a direct proxy for some ES (Table 3-4) due to unavailability of supporting data applicable for 1946. This means that the underlying factors considered in the SENCE method were not all incorporated in mapping such ES. Though it can be argued that the habitat type has a strong influence in delivering these ES, inclusion of other underlying data sets would have better reflected how such factors influence their delivery.

The use of multiple data sets as part of the SENCE method could also be a limitation in that such data sets have varying accuracy levels and scales. This could in some cases increase the degree of uncertainty, for example, in mapping the land erosion risk ES the underlying soil data set had a very small scale (1: 250 000) and hence the resulting output map showed less detail.

Another inevitable potential source of error was introduced during the conversion of the habitat layers from vector format into raster format (gridding). This is because, the shape of the habitat polygons (vector format) is converted into grid cells (pixels) in raster. In the process the integrity of the data is altered as small polygons are lost as this process allocates the pixel value to the habitat class that makes up the majority of the cell (Congalton, 1997). Also clipping data layers created artificial edge effects which could have potentially influenced estimation of ES values around the catchment boundaries. Steps like gridding were necessary as they enabled the application of the algorithms (ES values from the look up tables) used in the SENCE method. Besides it is much easier to analyse, compare large amounts of mapped data and present the maps in this format (Congalton, 1997, Swetnam, 2007, Taylor et al., 2000).

3.6.5 Accuracy assessment of the ecosystem service maps generated in this study

Currently, accuracy assessment in ES mapping is insufficiently addressed as approaches to test for accuracy are limited (Grêt-Regamey et al., 2015, Willemen et al., 2015). Vrebos et al. (2015a) for example considered stakeholder consultation as a way of validating ES maps and hence improving their accuracy. In this study, the current (2009) ES maps produced for the Scottish borders LUS pilot project were validated through local stakeholder consultation meetings, as well as review by Tweed Forum, a locally based participative catchment NGO. The local stakeholders provided ground based information on location and distribution of relevant ES in their area and this led to further refinement and improvement of these ES maps (Spray, 2014). Such stakeholders were also a potential reliable way of validating the generated historic ES maps, especially elderly stakeholders who have remained in these study areas for a long time. Unfortunately, this could not be achieved due to resource constraints. It can, however, be inferred that since historic ES mapping adopted the same methodology used to map the current ES in these sub catchments; validated by local stakeholders, the generated 1946 ES maps can be accepted as a good indication of ES delivery at that time.

3.7 Summary: Overall error budget from the data collection and processing stages

The figure below shows a collective presentation of the sources of error and uncertainty during the entire data collection and processing stages.

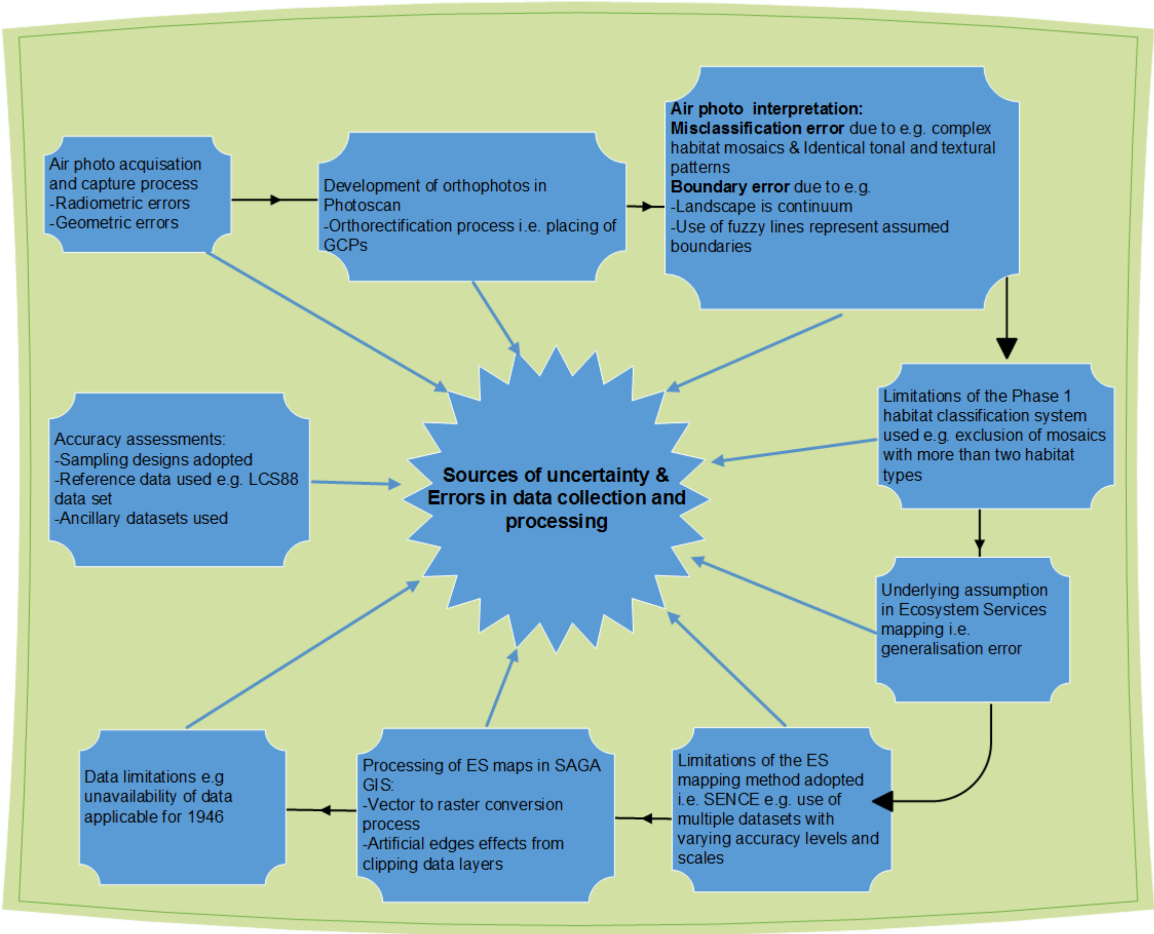


Figure 3-39: Data collection and processing error budget

Modified from Congalton (2001)

As illustrated in the figure above, possible sources of error were present in all the data collection procedures. As already discussed in the sections above, quality control measures were adopted to minimise these errors and accuracy assessments undertaken to assess the accuracy of the generated maps. Perhaps of fundamental importance is the question of what impact did these errors have on the intentions of this study i.e. assessing changes in ecosystem service delivery over time. Eigenbrod et al. (2010b) argue that the effect of the errors arising in ES mapping depends on the purpose and level of analysis of the generated maps. These authors suggest that if, like in this study, the aim of the study is to assess broad scale ES trends, patterns or to compare whether a certain area shows more or less of a certain ES type compared to the other, then ES maps derived from proxies such as habitat/LC/LU data can reliably and accurately provide such information.

3.8 Chapter summary

The aim of this chapter was to describe the data collection and processing procedures followed in this study. The chapter showed that three main stages of data collection and processing were followed. Firstly, air photos from the 1940s were used to reconstruct the landscape photo mosaics of the study catchments. This was followed by visual interpretation of these in GIS. Using the backdating approach, the current habitat map was edited during the interpretation process to derive the historic habitat maps for the study catchments. Lastly, the generated habitat maps were translated into historic ecosystem service supply maps using the SENCE method. These procedures were undertaken in order to detect changes in habitats and ecosystem services in the study catchments.

A discussion of sources of uncertainty and challenges encountered during the different stages of data collection and processing were presented. While quality control measures were followed to minimise some of the errors, there were inevitable inherent sources of uncertainty associated with spatial data, the fact that the landscape is a continuum rather than a series of discrete classes, as well as the subjectivity and confusion from visual interpretation of black and white air photos. Overall, however, accuracy assessment procedures followed showed that the generated photo mosaics, habitat and ecosystem services maps were highly accurate to meet the intentions of this study.

The next chapter presents the results from the analysis of the habitat and ecosystem service changes between the two dates in the study catchments focussing on (1) identifying the nature of change in both habitats and ecosystem services, (2) presenting the areal extent of these changes, and (3) assessing the spatial pattern of these changes.

4 Results

4.1 Chapter introduction

This chapter presents findings from the analysis of habitat and ecosystem services maps produced during the data collection and processing stages described in the previous chapter. The overall data analysis procedure was aimed at understanding the spatial changes that have occurred to habitats and ecosystem services in the Ale and Eddleston catchments as well as the spatial patterns and location of such changes. Ecosystem services change analysis in particular focussed on spatial assessment of the study catchments` capacities to supply selected ecosystem services and the changes that have occurred between 1946 and 2009, the two time periods which this study focusses on.

As discussed in chapter two (literature review), spatial assessment of ecosystem services provides spatial explicit information which among others could show areas of multiple ecosystem service provision and areas of conflict between ecosystem services, as well as opportunity areas where ecosystem service provision can be enhanced. In addition, mapping ecosystem services could also show degraded areas within a landscape which require restoration or protection in order to enhance ecosystem service supply. In this way spatial assessment of ecosystem services could inform evidence based decision making, environmental policy development and facilitate stakeholder engagement in environmental management.

In order to assess spatial changes in both habitats and ecosystem services in the two study catchments a series of steps, illustrated in the data analysis flow chart below (Figure 4-1) were done. Firstly, habitat change analysis was done and used as a basis for assessing spatial changes in ecosystem services in the study catchments. This is because habitats are noted to have a strong influence on ecosystem service delivery (Burkhard et al., 2012, Bolliger and Kienast, 2010), as they are primary landscape units providing ecosystem services. The classification of habitats significantly overlaps with that of ecosystems and have since been used to map ecosystem services as was done in the UK National Ecosystem Assessment (UK-National Ecosystem Assesment, 2011).

Section one of this chapter presents the results from habitat change analysis while the second section focusses on changes in ecosystem service delivery between 1946 and

2009. Each of these sections are aligned to the research questions posed in this study. Similarities and differences in the patterns of change in both catchments are identified.

As illustrated in the flow chart below, habitat maps were swapped between vector and raster formats to meet the preferred data formats of software used to analyse different aspects of change. The ecosystem services maps were analysed in raster format and data derived from them was further analysed in Microsoft Excel.

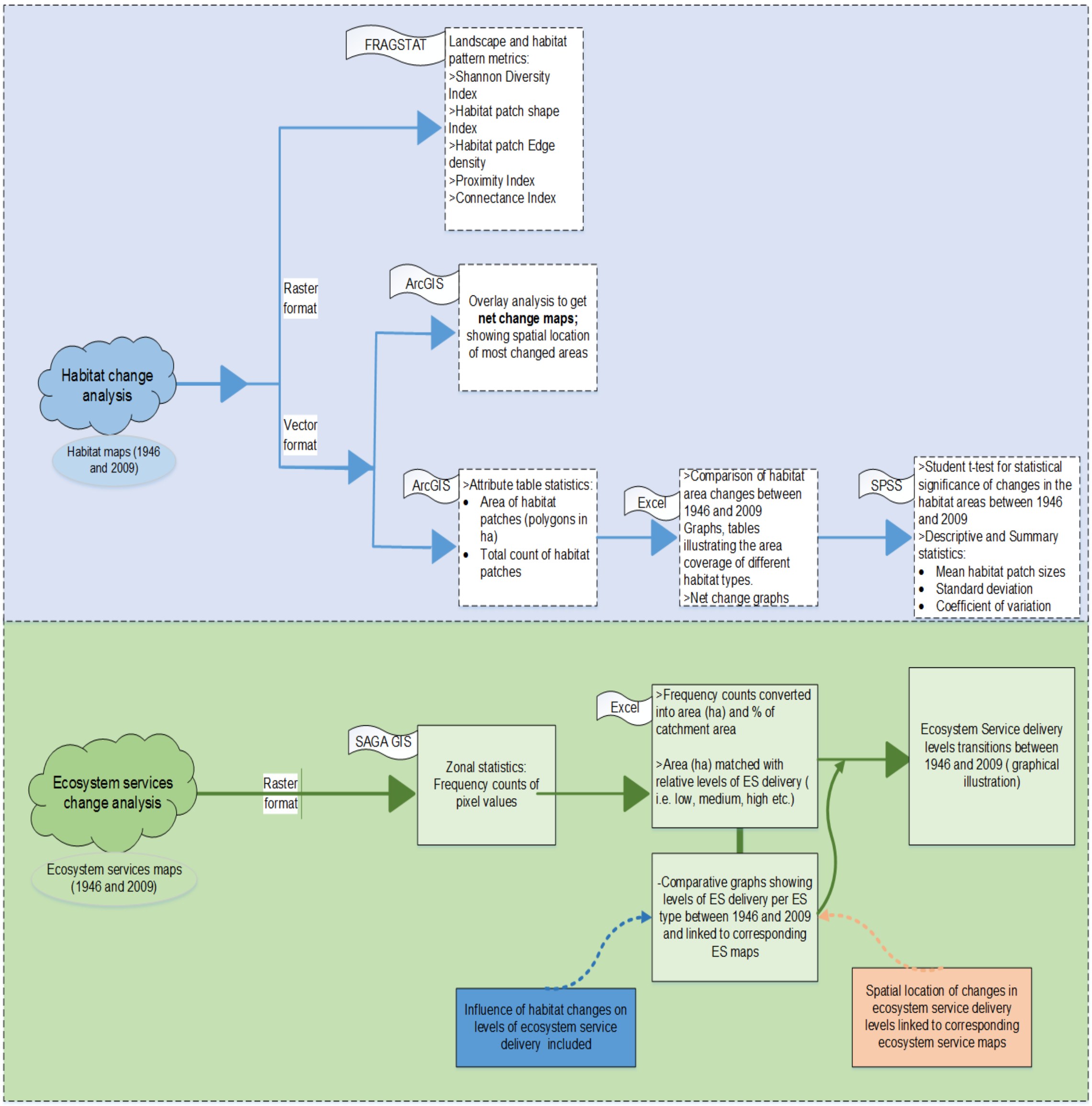


Figure 4-1: Data analysis flow chart

4.2 Section 1: Habitat changes in the Ale and Eddleston catchments between 1946 and 2009

This section presents the results from habitat change analysis, guided by the following sub research questions:

a. What is the historic state of habitats in the Ale and Eddleston catchments?
b. What spatial changes have occurred to habitats in these catchments between the 1940s and early 21st century?

To answer these research questions, firstly, the areal extent of habitats and changes to these between 1946 and 2009 are presented. The second section shows spatial locations within the study catchments where habitat changes occurred between these two dates as well as location of areas that remained unchanged. This is followed by an assessment of patterns of habitat change i.e. measures/indices of landscape and habitat fragmentation and diversity. This is of particular interest as ecosystem service delivery does not only depend on habitat type, habitat extent and spatial location but also on patterns of habitat changes which influence the configuration of habitat patches within the landscape, their connectivity and diversity. The calculations and habitat maps used to derive the summative graphs, tables and narrations presented in these sub sections are included in the appendices.

4.2.1 Areal extent of habitats in 1946 and 2009 in the study catchments

The area (ha) of habitat types found in the study catchments was computed from the 1946 and 2009 habitat maps (Appendix 10). The table below shows the area occupied by the main habitat types found in these catchments in 1946 and 2009 while Figure 4-2 and 4-3 are graphical illustrations on the area occupied by these habitat types. Major increases and decreases in habitat area are discussed following the table and the graphs and where applicable, the statistical significance of these changes is also included.

Table 4-1: Area (ha) occupied by habitat types in the Ale and Eddleston catchments in 1946 and 2009

	Ale catchment								Eddleston catchment							
	Year 1946		Year 2009						Year 1946		Year 2009					
Habitat type	Area (ha)	%	Area (Ha)	%	Net change in habitat area (ha)	Net change in area as percentage of individual habitat area	Net change of habitat area as percentage of total catchment area	Level of statistical significance	Area (ha)	%	Area (ha)	%	Net change in habitat area (Ha)	Net change in area as percentage of individual habitat area	Net change of habitat area as percentage of total catchment area	Level of statistical significance
Improved grassland	3031.67	17.77	4292.53	25.16	1260.86	41.59	7.39	***	1541.85	19.45	3373.55	42.56	1831.70	118.80	23.11	***
Coniferous woodland	147.09	0.86	3603.08	21.12	3455.99	2349.58	20.25	***	106.67	1.35	1018.88	12.85	912.21	855.17	11.51	**
Acid grassland - unimproved	712.77	4.18	1047.22	6.14	334.45	46.92	1.96	***	330.36	4.17	819.52	10.34	489.16	148.07	6.17	***
Dry dwarf shrub heath	139.49	0.82	399.89	2.34	260.40	186.68	1.53	**	9.20	0.12	45.81	0.58	36.62	398.14	0.46	–
Mixed woodland	273.72	1.60	321.24	1.88	47.52	17.36	0.28	***	351.93	4.44	398.41	5.03	46.48	13.21	0.59	***
Bracken	252.09	1.48	382.05	2.24	129.96	51.55	0.76	**	45.29	0.57	107.04	1.35	61.75	136.34	0.78	–
Built land	283.99	1.66	347.02	2.03	63.03	22.19	0.37	–	241.61	3.05	283.30	3.57	41.70	17.26	0.53	–
Gardens	1.63	0.01	10.95	0.06	9.32	571.36	0.05	*	3.99	0.05	20.92	0.26	16.94	424.78	0.21	***
Standing water	117.17	0.69	198.86	1.17	81.69	69.72	0.48	**	43.08	0.54	50.47	0.64	7.39	17.15	0.09	–
Other exposure i.e. caravan sites, refuse tip	0.00	0.00	11.84	0.07	11.84	100.00	0.07	n/a	6.04	0.08	15.45	0.19	9.41	155.79	0.12	n/a
Marsh/marshy grassland	386.56	2.27	415.03	2.43	28.47	7.36	0.17	***	651.48	8.22	256.95	3.24	-394.53	-60.56	-4.98	***
Fen-valley mire	76.22	0.45	116.37	0.68	40.15	52.68	0.24	–	92.87	1.17	77.82	0.98	-15.05	-16.21	-0.19	–
Dry heath/acid grassland	309.53	1.81	730.57	4.28	421.04	136.02	2.47	*	86.90	1.10	65.51	0.83	-21.38	-24.61	-0.27	–
Bogs	4163.36	24.40	922.60	5.41	-3240.76	-77.84	-18.99	***	1679.87	21.2	492.09	6.21	-1187.78	-70.71	-14.98	*
Cultivated land - arable	2857.99	16.75	2497.12	14.63	-360.87	-12.63	-2.11	***	819.22	10.34	152.07	1.92	-667.16	-81.44	-8.42	*
Poor semi-improved grassland	730.42	4.28	386.68	2.27	-343.74	-47.06	-2.01	***	431.08	5.44	96.47	1.22	-334.61	-77.62	-4.22	*
Acid grassland - semi-improved	924.81	5.42	639.94	3.75	-284.87	-30.80	-1.67	***	609.39	7.69	429.78	5.42	-179.61	-29.47	-2.27	–
Wet dwarf shrub heath	322.44	1.89	108.93	0.64	-213.51	-66.22	-1.25	–	148.16	1.87	0.00	0.00	-148.16	-100.00	-1.87	**
Wet heath/acid grassland	1489.42	8.73	365.41	2.14	-1124.01	-75.47	-6.59	**	374.44	4.72	20.90	0.26	-353.54	-94.42	-4.46	*
Broadleaved woodland	431.60	2.53	114.14	0.67	-317.46	-73.55	-1.86	***	131.01	1.65	52.52	0.66	-78.49	-59.91	-0.99	**
Scrub	167.58	0.98	60.70	0.36	-106.88	-63.78	-0.63	***	86.63	1.09	31.18	0.39	-55.45	-64.01	-0.70	***
Neutral grassland	151.15	0.89	55.34	0.32	-95.81	-63.39	-0.56	**	35.79	0.45	11.37	0.14	-24.42	-68.23	-0.31	–
Flush and spring - acid/neutral flush	79.81	0.47	35.86	0.21	-43.95	-55.07	-0.26	**	96.23	1.21	70.07	0.88	-26.16	-27.19	-0.33	–
Excavation site/Quarry	12.85	0.08	0.00	0.00	-12.85	-100.00	-0.08	n/a	3.48	0.04	36.50	0.46	33.02	950.06	0.42	n/a
Total catchment area	**17063.4**	**100**	**17063.4**	**100**	**0**		**0**		**7926.6**	**100**	**7926.6**	**100**	**0**		**0**	

Legend:

	Habitat types that increased in area
	Habitat types that decreased in area
n/a	Not possible to test statistically as number of counts was too small
*	p<0.05
**	p<0.01
***	p<0.001
–	Statistically insignificant

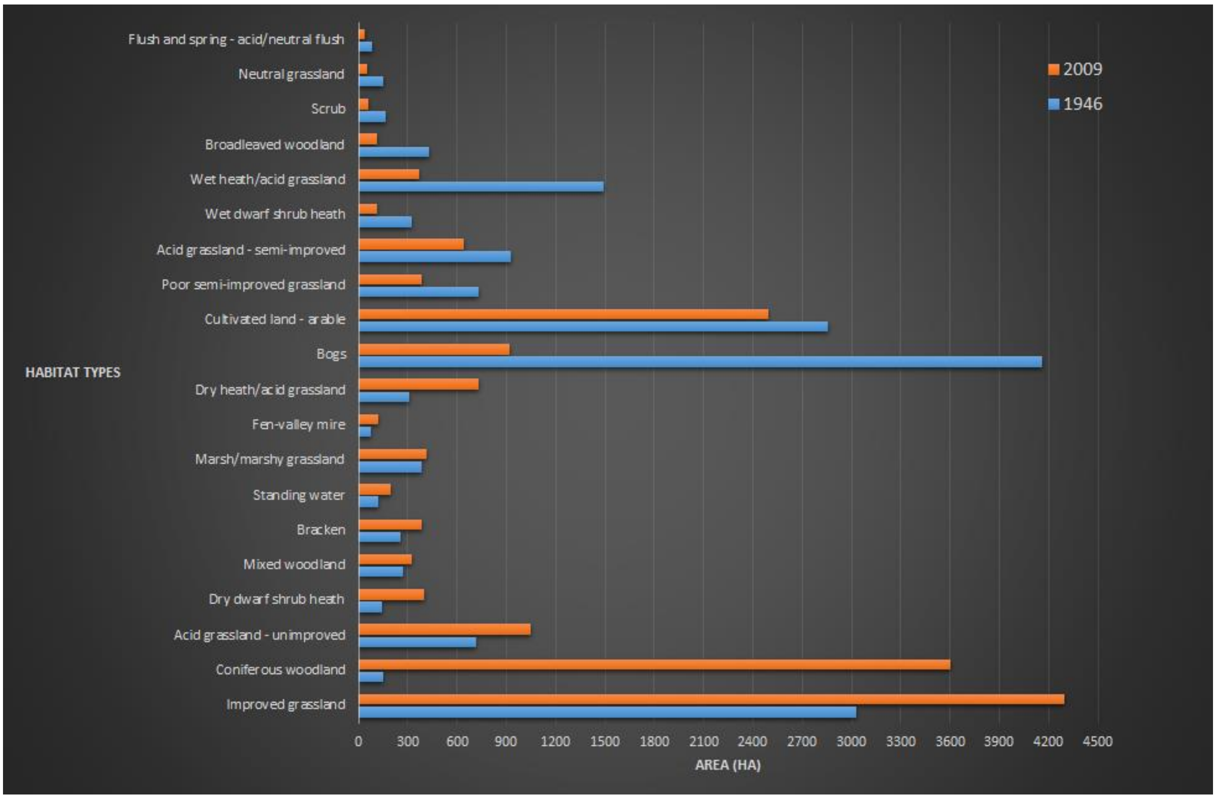


Figure 4-2: Area occupied by main habitat types in the Ale in 1946 and 2009

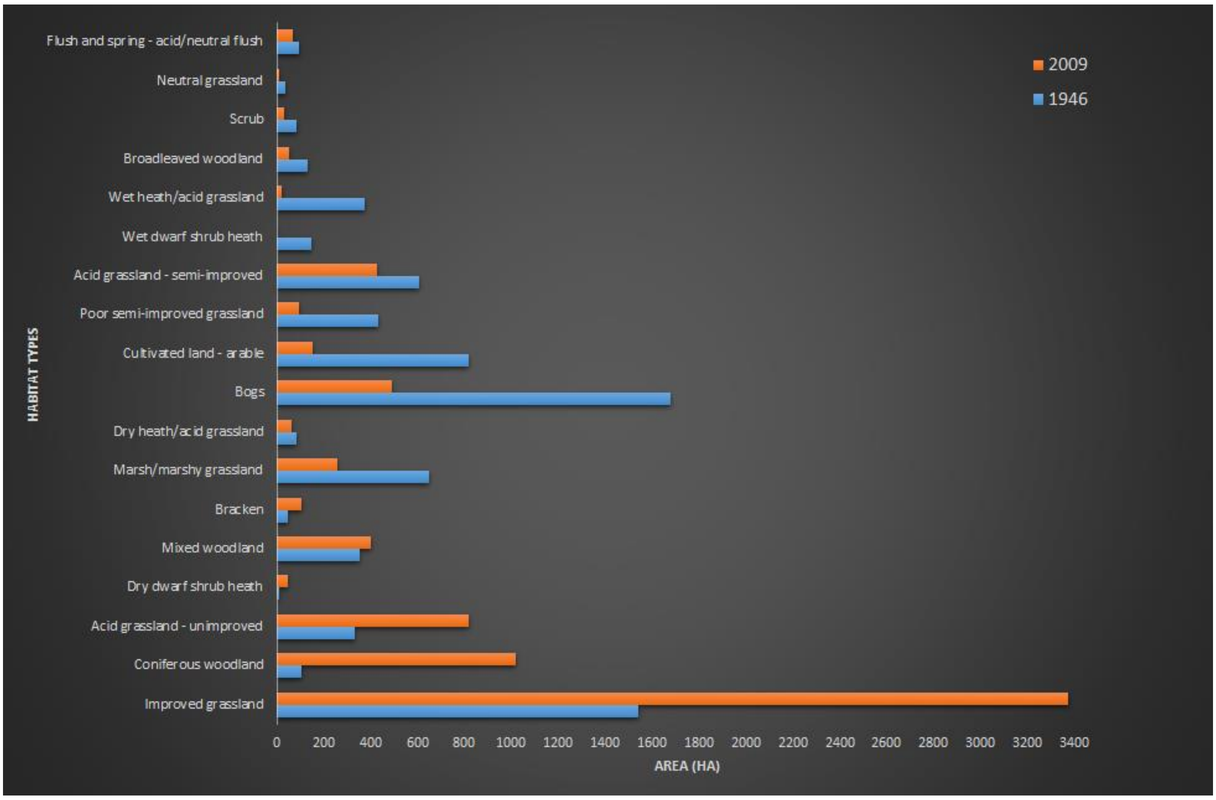


Figure 4-3: Area occupied by main habitat types in the Eddleston in 1946 and 2009

4.2.1.1 Habitat types that increased in area (ha)

Habitats types that increased in area between the two dates in both catchments were: improved grassland, coniferous woodland plantations, unimproved acid grassland, dry dwarf shrub heath, mixed woodland plantations, bracken, gardens, built land and standing water. Other increases recorded in either of the catchments include: marshy grassland, dry heath/acid grassland mosaics and fen valley mires in the Ale catchment and excavation sites in the Eddleston catchment. Area changes in each of these habitat types between the two dates are in turn discussed below.

Improved grassland

As illustrated in the table above (Table 4-1), in both years, improved grassland was the most widespread and prevalent habitat type in both catchments. In the Ale catchment, it was widespread in the mid and lower catchment, occupying about 18 % (3000 ha) of the total catchment area in 1946. It recorded a statistically significant increase ($t=24.458$, $p<0.001$) to account for about 25% (about 4300ha) of the total catchment area in 2009.

Similarly, in the Eddleston catchment, improved grassland was common in the valley floor in 1946 as it accounted for about 20% (approximately 1500 ha) of the total catchment area. In 2009, the area under this habitat type increased to more than double the area it occupied in 1946 (statistically significant) to cover about 43% (about 3400 ha) of the total catchment area and hence the most dominant habitat type in 2009 ($t=15.714$, $p<0.001$).

Coniferous woodland plantations

Coniferous woodland plantations were very limited in extent in 1946 in both catchments. For example, few plantations/patches, accounting for about 0.9% (about 150 ha) of the total Ale catchment area, were present in the lower and mid catchment areas of the Ale and non-existent in the upper catchment in 1946. In the Eddleston catchment, coniferous woodland plantations were also non-existent in the uplands in 1946 and only covered small areas within the valley floor; occupying about 1.4% (about 100 ha) of the total catchment area.

By 2009, coniferous woodland plantations had significantly increased in both catchments. In the Ale, coniferous woodland plantations spread throughout the catchment, with their highest dominance in the upper catchment where it covered extensive areas, accounting for about 21% (3600ha) of the total catchment area ($t=6.744$, $p<0.001$). In the Eddleston, coniferous woodland plantations also increased to occupy about 13% (about 1000 ha) of the total catchment area ($t=3.470$, $p<0.01$) and thus became a widespread habitat type in this catchment; especially on the western uplands of the Eddleston valley.

The pictures[13] below show some areas within the Ale catchment that were taken over by coniferous woodland plantations in 2009.

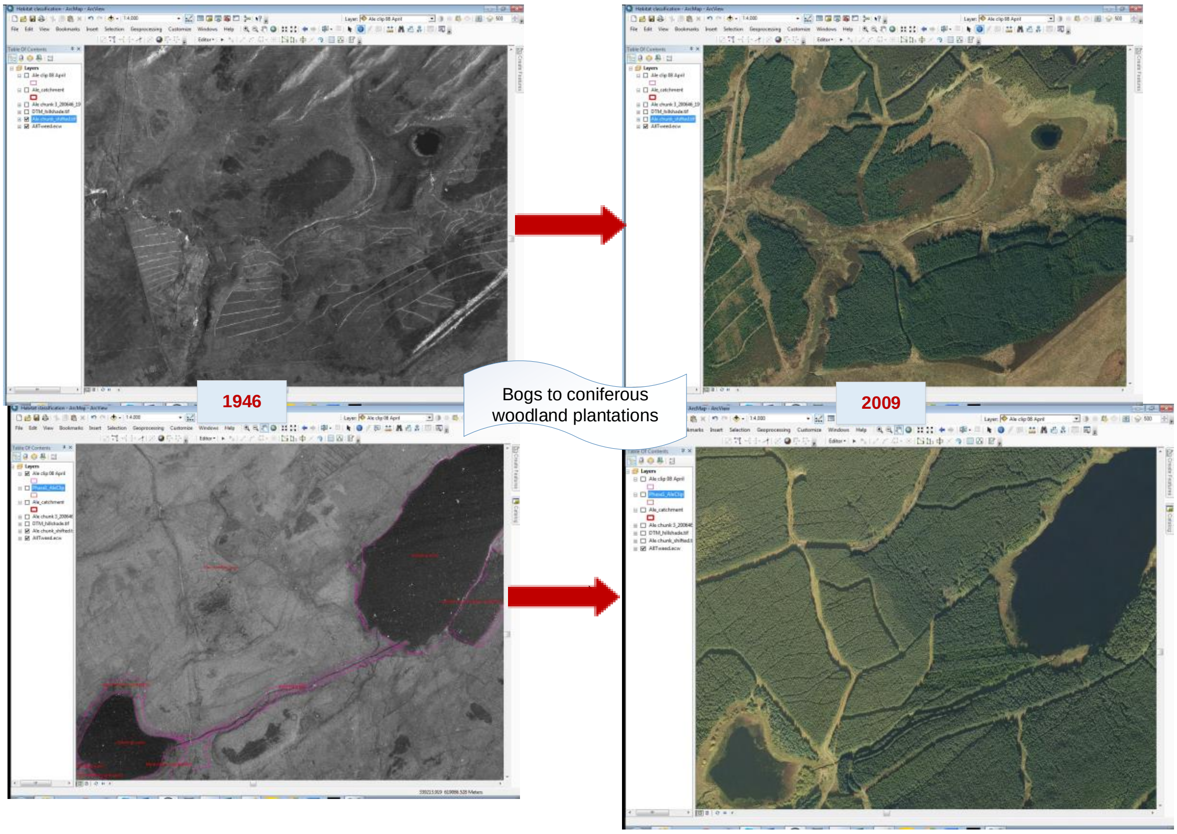


Figure 4-4: Increase in coniferous woodland plantations in the uplands of the Ale catchment

In figure 4-4, the pictures on the left show the modified bogs around the reservoirs in the uplands of the Ale catchment in 1946. By 2009, this area was dominated by coniferous woodland plantations as illustrated in the pictures on the right.

Unimproved acid grassland

The area under unimproved acid grassland increased (statistically significant) in both catchments. In 1946, the total area occupied by unimproved grassland was about 4.2%

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[13] 1946 pictures shown in this chapter are an extract from the 1946 photo mosaics developed in this study and the 2009 colour aerial photography are an extract from the 2009 ©getmapping colour aerial photography for the Ale and Eddleston catchments

(713 ha) of the Ale total catchment area while it also covered approximately 4.2% (330 ha) of the Eddleston total catchment area. By 2009, unimproved acid grassland had increased to account for 6% (1000ha) of the Ale catchment total catchment area ($t=7.039$, $p<0.001$) and 10% (800 ha) of the Eddleston total catchment area ($t=7.243$, $p<0.001$).

Dry dwarf shrub heath and dry heath/acid grassland mosaics

Area under dry dwarf shrub heath increased in both catchments between the two dates while area under dry heath/acid grassland recorded an increase in the Ale and decreased in the Eddleston catchment. The total Ale catchment area under dry dwarf shrub heath more than doubled (statistically significant) from about 0.8% (140 ha) in 1946 to about 2% (400 ha) in 2009 ($t=3.951$, $p<0.01$). Also in the Ale catchment, area under dry heath/acid grassland mosaics substantially increased (statistically significant) from about 2% (309 ha) in 1946 to about 4% (730 ha) in 2009 ($t=3.068$, $p<0.05$). In the Eddleston catchment, though statistically insignificant, dry dwarf shrub heath accounted for 0.1% (about 9 ha) of the total catchment area in 1946 and it increased to cover about 0.6% (46 ha) of the total catchment area in 2009.

Mixed woodland

In both catchments, mixed woodland plantations increased between the two dates. In the Ale catchment, mixed woodland plantation patches occupied about 1.6 % (270 ha) of the total catchment area in 1946. These mixed woodland plantations were mainly located in the lower and mid catchment; along roads and around fields. In 2009, mixed woodland plantations increased significantly to form riparian woodland along rivers and streams occupying an area of about 2% (321 ha) ($t=6.666$, $p<0.001$) of the total catchment area.

In the Eddleston catchment mixed woodland plantations occupied about 4.4 % (351 ha) in 1946 and increased significantly, in 2009 to occupy approximately 5% (about 398 ha) of the total catchment area ($t=7.096$, $p<0.001$). Mixed woodland plantations were the most dominant woodland type in 1946 compared to broadleaved or coniferous woodlands in the Eddleston catchment.

Bracken

Between these two time periods, both catchments recorded an increase in the presence of bracken habitat patches. The Ale catchment recorded a statistically significant increase from about 1.5% (250 ha) in 1946 to about 2% (380 ha) of the total catchment area in 2009 ($t=3.945$, $p<0.01$). The Eddleston recorded a (statistically insignificant) increase from 0.6% (45 ha) in 1946 to 1.2% (about 100 ha) of the total catchment area in 2009.

Built land

Both catchments recorded slight (statistically insignificant) increases in the area under built land (roads, settlements etc.) between the two time periods. In the Ale catchment, built land increased from about 1.7% (284 ha) of the total catchment area in 1946 to about 2% (about 347 ha) in 2009. The spatial location of these built up areas did not change though roads were expanded and a few built up areas were introduced in the catchment such as the caravan site located on the north east end of the lower catchment (appendix 10a).

The Eddleston catchment also recorded an increase (statistically insignificant) in built land from about 3% (about 242 ha) in 1946 to about 3.6% (about 283ha) of the total catchment area in 2009, mainly emanating from the expansion of roads and the two main settlement areas found in this catchment i.e. Peebles and the Eddleston village (Figure 4-5). For example, the 2009 picture (on the right in the figure below) shows that the Eddleston village spread towards the North East, replacing cultivated arable fields and improved grassland. Also a few built up areas such as the Millennium farm were also introduced by 2009 in this catchment.

Figure 4-5: Eddleston village in 1946 and 2009

Gardens

Both catchments recorded an increase in area under gardens. In 1946, few gardens could be identified in both catchments i.e. 0.01% (about 1.6 ha) of the total Ale catchment area and 0.05% (about 4 ha) of the Eddleston total catchment area. A statistically significant increase in the area of gardens was recorded in 2009, accounting for 0.06% (about 11 ha) of the Ale catchment (t=3.670, $p<0.05$) and about 0.3% (about 21 ha) of the Eddleston catchment (t=8.546, $p<0.001$).

Rivers, streams and reservoirs

The areal extent of rivers, streams and reservoirs in both catchments between the two time periods have remained similar. For example, the main reservoirs found in the upper and mid catchment areas of the Ale have remained the same between two time periods save for the Almoor Loch, which increased its surface area as illustrated in Figure 4-6. Also in 2009 there was an increase in the number of small ponds in both catchments which were very few in 1946. This contributed to the overall increase in the area under standing water from 0.7% (about 117 ha) of the total Ale catchment area in 1946 to about 1.2% (about 200ha) in 2009 (t=3.345, $p<0.01$). Though statistically insignificant the Eddleston catchment also recorded an increase in area under standing water from about 0.5% (about 43ha) of the total catchment area in 1946 to about 0.6% (about 50 ha) in 2009.

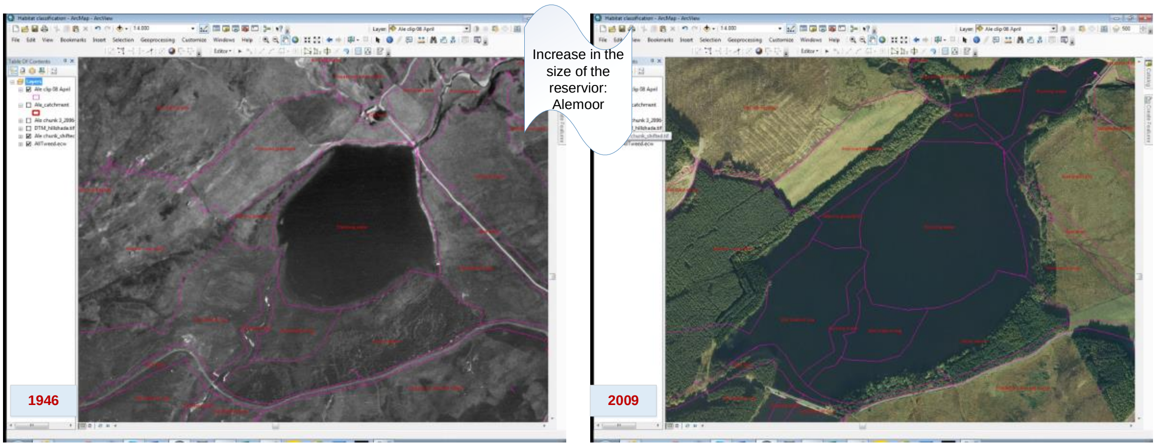

Figure 4-6: Increase in the surface area of the Almoor Loch

Marshy grassland in the Ale catchment

The Ale catchment recorded a statistically significant increase in the area under marshy grassland from 2.3% (387 ha) of the catchment area in 1946 to occupy about 2.4% (415 ha) of the total catchment area in 2009 (t=5.121, $p<0.001$).

Fen valley mire in the Ale catchment

Though statistically insignificant, area under fen valley mires in the Ale catchment increased from about 0.5% (76 ha) of the total catchment area in 1946 to about 0.7% (116 ha) in 2009.

Excavation sites in the Eddleston catchment

The Eddleston catchment recorded an increase in excavation sites from about 0.04% (3.5 ha) of the total catchment area in 1946 to about 0.5% (about 37 ha) in 2009. This increase resulted from the introduction of new excavation sites such as the Cowieslinn quarry and open cast site (Figure 4-7) recorded in 2009. In contrast, a few excavation sites i.e. dug up pits limited in extent were identified in the lower catchment areas of the Ale in 1946, covering about 0.08% (about 13ha) of the total Ale catchment area but these were not recorded in 2009.

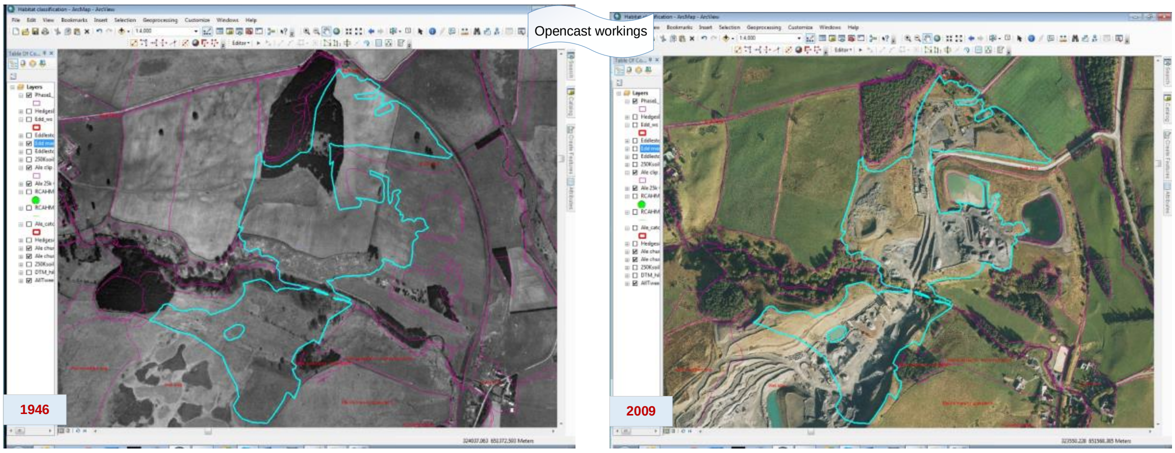

Figure 4-7: Example of new quarry site present in 2009

As shown in figure 4-7, the picture on the left shows how the current quarry site area was in 1946. Habitat types such as semi- improved grassland areas and mixed woodland plantations were replaced by this site.

4.2.1.2 Habitat types that decreased in areal extent

Habitats types that decreased in area between the two dates in both catchments were: bogs, cultivated land, poor semi-improved grassland, semi-improved acid grassland, wet dwarf shrub heath, wet heath/acid grassland mosaics, broadleaved woodland, hedgerows, scrub, neutral grassland and flush/springs. Other decreases recorded in either of the catchments include: dry heath/acid grassland mosaics, fen valley mires and marshy grassland areas in the Eddleston catchment. The changes in area for each of these habitat types are in turn discussed below.

Bogs

Both catchments recorded a massive reduction in the area under bogs (inclusive of wet bogs, blanket bogs, dry and wet modified bogs) as they were mostly taken over by coniferous woodland plantations. In the Ale catchment, bogs accounted for nearly 24% (4100 ha) of the total catchment area in 1946. These were reduced to about 5% (about 900 ha) in 2009 (statistically significant: $t=5.043$, $p<0.001$). Similarly, in the Eddleston catchment, bogs accounted for about 21% (about 1600 ha) of the total catchment area in 1946 and these were reduced to about 6% (about 500 ha) in 2009 ($t=2.217$, $p<0.05$).

Cultivated land- arable

Both catchments recorded a decrease in the area under arable land. In the Ale catchment, cultivated land accounted for about 17% (about 2900 ha) of the total catchment area in 1946. By 2009, area under arable land decreased significantly to account for about 14.6% (about 2 500 ha) of the total Ale catchment area ($t=4.295$, $p<0.001$). Despite this decrease, cultivated land remained the most dominant habitat type in the lower and mid catchment areas of the Ale in 2009, where it was mainly interspersed with improved grassland areas. The Eddleston catchment recorded a striking reduction in the area under cultivated land from a coverage of about 10% (819 ha) of the total catchment area in 1946 to about 2% (152ha) in 2009 ($t=2.761$, $p<0.05$). Most of these arable land areas within the Eddleston valley floor were taken over by improved grassland.

Poor semi-improved grassland

Both catchments recorded a significant reduction in the area under poor semi-improved grassland. In the Ale catchment, a decrease from about 4.3. % (730 ha) of the total catchment area in 1946 to 2.3% (386 ha) in 2009 ($t=10.021$, $p<0.001$) was observed. In the Eddleston catchment a marked decrease from nearly 5.4 % (430 ha) of the total catchment area in 1946 to about 1.2% (97 ha) in 2009 ($t=2.428$, $p<0.05$) was recorded. These areas were mainly taken over by improved grassland areas.

Semi-improved acid grassland

Both catchments recorded a decrease in the area under semi-improved acid grassland. In 1946, the total Ale catchment area occupied by semi-improved acid grassland patches was about 5 % (925 ha). In 2009, the area under semi-improved acid grassland in this catchment decreased to occupy about 4% (640 ha) of the Ale total catchment area (statistically significant: $t=8.762$, $p<0.001$). In the Eddleston, semi-improved acid grassland occupied 8% (about 600 ha) of the total catchment area in 1946. In 2009, though statistically insignificant the area under semi-improved acid grassland reduced to occupy about 5% (430 ha) of the total catchment area.

Wet dwarf shrub heath and wet heath/acid grassland mosaics

Both catchments recorded a decrease in area under wet dwarf shrub heath and wet heath/acid grassland mosaics. In the Ale catchment, though statistically insignificant the area under wet dwarf shrub heath declined from about 1.9% (322 ha) of the total catchment area in 1946 to about 0.6% (108 ha) in 2009. Also in this catchment, area under wet heath/acid grassland mosaics significantly reduced from about 9% (1490 ha) of the total Ale catchment area in 1946 to about 2% (365ha) in 2009 ($t=3.052$, $p<0.01$).

In the Eddleston catchment, wet dwarf shrub heath accounted for about 1.2% (148 ha) of the total catchment area in 1946 and it was not recorded in 2009; suggestive of its decline to very low area coverage. Also in this catchment, both wet and dry heath/acid grassland reduced from 6% (about 460 ha) of the total Eddleston catchment area in 1946 to about 1 % (87 ha) in 2009 ($t=4.020$, $p<0.01$).

Broadleaved woodland

Both catchments recorded a decrease in broadleaved woodland (inclusive of broadleaved parklands, plantations and semi-natural occurrences). In the Ale catchment, broadleaved woodland plantations occupied about 2.5% (about 430 ha) of the total catchment area in 1946. In 2009, there was a significant decrease in the area occupied by broadleaved woodland plantations in the Ale catchment to about 0.7% (114 ha) ($t=4.624$, $p<0.001$). The Eddleston catchment also recorded a significant decrease in the area under broadleaved woodland plantations from about 1.7% (130 ha) of the total catchment area in 1946 to about 0.7% (53 ha) in 2009 ($t=3.420$, $p<0.01$).

Hedgerows

Both catchments recorded a decrease in the concentration of hedgerows around fields and along roads. In 1946, hedgerows were highly concentrated in the lower and mid catchment areas of the Ale and in the Eddleston valley floor as field boundaries and also concentrated along roads. In 2009, the concentration of hedgerows had declined with the total length[14] of hedgerows in the Ale catchment decreasing by approximately 37% from 316.59 km in 1946 to approximately 200.82km. In the Eddleston catchment hedgerows decreased by approximately 38% from 41.46 km in 1946 to approximately 25.56 km in 2009. Figures 4-8 and 4-9 below give an illustration on some of hedgerows that were present in 1946 but had been removed by 2009.

Figure 4-8: Illustration of removal of hedgerows in the Ale catchment

The picture on the left with the red pins in Figure 4-8, above, shows a network of fields lined up by hedgerows in the Ale catchment, the same fields in 2009 are shown in the picture on the right with hedgerows removed.

Figure 4-9: Removal of hedgerows between 1946 and 2009 in the Eddleston catchment

The red pins on the black and white picture on the left in the figure above show fields that were demarcated using hedgerows in 1946 in the Eddleston catchment. These were later removed and the same field (picture on the right) had been merged into one by 2009.

[14] The changes in hedgerows was expressed in terms of length (km) rather than area.

Scrub

Both catchments recorded a significant decrease in the area under scrub between the two time periods. In 1946, scrub accounted for about 1% (168 ha) of the Ale catchment and this decreased to about 0.4% (60 ha) in 2009 ($t= 5.501$, $p<0.001$). In the Eddleston catchment, area under scrub reduced from about 1 % (87 ha) of the total catchment area in 1946 to about 0.4% (30 ha) in 2009 ($t= 4.457$, $p<0.001$).

Neutral grassland

Both catchments recorded a decrease in the area under neutral grassland. In the Ale catchment, neutral grassland reduced from about 0.9% (150 ha) of the total catchment area in 1946 to about 0.3% (55 ha) in 2009 ($t=3.519$, $p<0.01$). In the Eddleston catchment; though statistically insignificant, neutral grassland also decreased from about 0.5% (36 ha) of the total catchment area in 1946 to about 0.1% (11 ha) in 2009.

Flush and Springs

Both catchments recorded a decrease in the area under flush and springs. The Ale catchment recorded a significant decrease from about 0.5% (80 ha) of the total catchment area in 1946 to about 0.2% (36 ha) in 2009 ($t=4.623$, $p<0.01$). In the Eddleston catchment, though statistically insignificant area under flush and springs decreased from about 1.2% (96 ha) of the total catchment area in 1946 to about 0.9 % (70 ha) in 2009.

Fen valley mires in the Eddleston catchment

The Eddleston catchment recorded a statistically insignificant decrease in area under fen valley mires. In 1946, fen valley mires accounted for about 1.2% (90 ha) of the total catchment area and this reduced to about 0.9% (78 ha) in 2009.

Marshy grassland in the Eddleston catchment

In contrast to the Ale catchment, the Eddleston also recorded a decrease in area under marshy grassland. In 1946, marshy grassland areas were prominent along seepage lines at the bottom of slopes and along rivers and streams within the Eddleston valley floor accounting for about 8% (652 ha) of the total catchment area. In 2009, marshy grassland areas significantly decreased to account for about 3% (260 ha) of the total catchment area ($t=5.914$, $p<0.001$).

4.2.1.3 Net percentage changes in habitat area between 1946 and 2009

The figure below shows percentage changes in habitat area between 1946 and 2009 in study catchments.

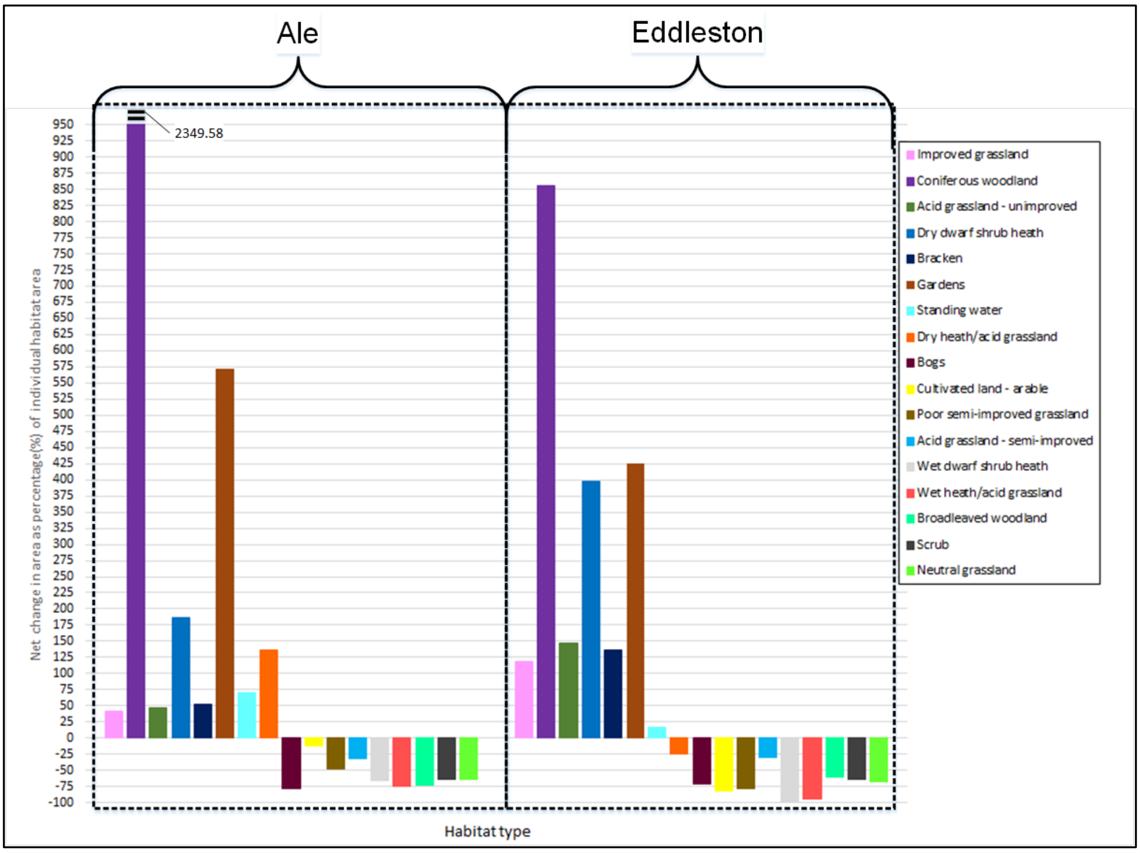

Figure 4-10: Percentage increases and decreases in habitat area between 1946 and 2009 in the Ale and Eddleston catchments

Figure 4-10 above shows that coniferous woodland recorded the highest habitat area percentage increase of over 1000% in the Ale catchment and 855% in the Eddleston catchment. As discussed in the previous section, coniferous woodland increased significantly between 1946 and 2009 to occupy great proportions of total catchment areas in 2009 and became the second most dominant habitat type in these catchments after improved grassland. Area under gardens recorded the second highest percentage increase of about 575% in the Ale catchment and 420% in the Eddleston catchment. Despite these marked percentage increases, gardens were of less prominence in these catchments in 2009 as they accounted for small proportions of total catchment areas.

Dry dwarf shrub heath and dry heath/acid grassland recorded percentage increases of over 100% in the Ale catchment. The Eddleston catchment recorded a much higher percentage increase of nearly 400% for area under dry dwarf shrub heath and a percentage decrease of about 25% in area under dry heath/acid grassland mosaics. Dry dwarf shrub heath increased to be among the common habitat types found in these catchments in 2009, accounting for significant total catchment areas. Likewise, dry heath/acid grassland was also common in the Ale catchment in 2009 but less prominent in the Eddleston during the same period as indicated by its decrease in area.

Also in the Ale catchment, relatively high percentage increases were recorded for improved grassland (42%), unimproved acid grassland (47%), bracken (52%) and standing water (70%). In the Eddleston catchment, these habitat types recorded a much higher percentage increase of 118% for improved grassland, 148% for unimproved acid grassland, 136% for bracken and 17% for standing water compared. However, given the difference in the catchment sizes the area occupied by these habitat types was larger in the Ale catchment; which is twice as big as the Eddleston. Improved grassland remained the most dominant habitat type in 2009 in both catchments alongside coniferous woodland, accounting for significant total catchment areas (Table 4-1). It, however, recorded a small percentage increase in the Ale catchment compared to the Eddleston catchment. Improved grassland was already a dominant habitat type in these catchments in 1946 and it further increased to remain so in 2009. Unimproved acid grassland also increased in 2009 to be among the common habitat types in both catchments. Bracken and standing water also recorded high percentage increases but these accounted for small total catchment areas in 2009 and were among the least prominent habitat types in these catchments.

On the other hand, bogs, wet heath/acid grassland and broadleaved woodland recorded very high percentage decreases in both catchments in 2009. A greater proportion of catchment areas under bogs in 1946 declined by about 80% in the Ale catchment and by about 70% in the Eddleston catchment. Wet heath/acid grassland mosaics also recorded a high percentage decrease of about 75% in the Ale catchment and over 90% in the Eddleston catchment in 2009 as these were also taken over by coniferous woodland plantations in the uplands. Consequently, these habitat types were less dominant in 2009, accounting for small catchment areas compared to their coverage in 1946. Broadleaved woodland plantations also recorded a significant percentage decrease of about 74% and 60% in the Ale and Eddleston respectively. However, this habitat type was not among the most dominant habitat types in these catchments in 1946 and neither was it in 2009 as it accounted for small total catchment areas in both time periods.

Another notable percentage decrease in both catchments was the area under cultivated land. Cultivated land recorded a small percentage decrease of about 13% in the Ale catchment while it recorded a marked percentage reduction of about 80% in the Eddleston catchment. Despite its low percentage decrease in the Ale catchment, this habitat type remained among the most dominant habitat types in this catchment in 2009 together with coniferous woodland and improved grassland as it accounted for a significant proportion of the total catchment area both in 1946 and 2009. In contrast, in the Eddleston catchment cultivated land was among the least common habitat types in 2009 opposed to its dominance in 1946.

Other recorded percentage decreases in both catchments were in area under poor semi-improved grassland, wet dwarf shrub heath, scrub and neutral grassland. The Eddleston catchment recorded a much higher percentage decrease in both poor semi-improved grassland (78%) and wet dwarf shrub heath (100%) compared to the Ale catchment which recorded a percentage decrease of 47% and 66% respectively for these habitat types. In both catchments, these habitat types became less prominent in 2009 compared to 1946 following the reduction in the total catchment areas occupied by these. Scrub and neutral grassland recorded percentage decreases of over 60% in both catchments. However, both these habitat types were less common in both catchments as they accounted for small catchment proportions of less than 1% of total catchment areas in both time periods.

4.2.2 Habitat transitions between 1946 and 2009 in the study catchments

The tables below show the habitat transitions and changes between 1946 and 2009 in the study catchments.

Table 4-2: Habitat transitions between 1946 and 2009 in the Eddleston catchment

1946 habitat type and area (ha)		Habitat transitions by 2009 (>10% of baseline area)
Improved grassland (1541.85 ha)	1247.94ha (81%)	Improved grassland
Coniferous woodland plantation (106.67 ha)	85.41ha (80%)	Coniferous woodland plantation
	18.3ha (17%)	Mixed woodland plantation
Acid grassland - unimproved (330.36 ha)	110.08ha (33%)	Improved grassland
	92.88ha (28%)	Acid grassland - unimproved
	45.4ha (14%)	Acid grassland - semi-improved
Dry dwarf shrub heath (9.20 ha)	9.2ha (100%)	Dry dwarf shrub heath - acid
Mixed woodland plantation (351.93 ha)	206.58ha (59%)	Mixed woodland plantation
	38.05ha (11%)	Broadleaved woodland plantation
	36.15ha (10%)	Coniferous woodland plantation
Bracken (45.29 ha)	33.32ha (74%)	Bracken
	8.32ha (18%)	Coniferous woodland plantation
Built land (241.61 ha)	212.77ha (88%)	Built land
	25.04ha (10%)	Improved grassland
Gardens (3.99 ha)	3.09ha (77%)	Gardens
	0.6ha (15%)	Mixed woodland plantation
Standing water (43.08 ha)	42.7ha (99%)	Standing water
Marsh/marshy grassland (651.48 ha)	331.37ha (51%)	Improved grassland
	161.25ha (25%)	Marsh/marshy grassland
	84.02ha (13%)	Acid grassland - semi-improved
Fen-valley mire (92.87 ha)	70.58ha (76%)	Fen - valley mire
	22.29ha (24%)	Improved grassland
Dry heath/acid grassland (86.90 ha)	51.49ha (59%)	Coniferous woodland plantation
	15.81ha (18%)	Improved grassland
	14ha (16%)	Bracken
Bogs (1679.87 ha)	502.43ha (30%)	Bogs
	433.28ha (26%)	Coniferous woodland plantation
	404.09ha (24%)	Acid grassland - unimproved
	180.06ha (11%)	Improved grassland
Cultivated land - arable (819.22 ha)	720.62ha (88%)	Improved grassland
Poor semi-improved grassland (431.08 ha)	297.65ha (69%)	Improved grassland
	70.7ha (16%)	Poor semi-improved grassland
Acid grassland - semi-improved (609.39 ha)	361.36ha (59%)	Improved grassland
	110ha (18%)	Acid grassland - unimproved
	79.52ha (13%)	Acid grassland - semi-improved
Wet dwarf shrub heath (148.16 ha)	36.44ha (25%)	Acid grassland - unimproved
	33.83ha (23%)	Coniferous woodland plantation
	28.54ha (19%)	Bracken
	21.05ha (14%)	Wet dwarf shrub heath - acid
	15.16ha (10%)	Acid grassland - semi-improved
Wet heath/acid grassland (374.44 ha)	104.28ha (28%)	Acid grassland - semi-improved
	69.91ha (19%)	Acid grassland - unimproved
	59.1ha (16%)	Wet heath/acid grassland
	51.2ha (14%)	Improved grassland
	49.21ha (13%)	Coniferous woodland plantation
Broadleaved woodland plantation (131.01 ha)	84.69ha (65%)	Mixed woodland plantation
	20.34ha (16%)	Improved grassland
Scrub (86.63 ha)	28.71ha (33%)	Improved grassland
	27.62ha (32%)	Mixed woodland plantation
	14.83ha (17%)	Coniferous woodland plantation
Neutral grassland (35.79 ha)	14.22ha (40%)	Improved grassland
	11.6ha (32%)	Mixed woodland plantation
	5.96ha (17%)	Broadleaved woodland plantation
Flush and spring - acid/neutral flush (96.23 ha)	71.1ha (74%)	Flush and spring - acid/neutral flush
Excavation site/Quarry (3.48 ha)	2.19ha (63%)	Improved grassland
	0.74ha (21%)	Scrub
	0.55ha (16%)	Mixed woodland plantation

Table 4-3: Habitat transitions between 1946 and 2009 in the Ale catchment

1946 habitat type and area (ha)		**Habitat transitions by 2009 (>10% of baseline area)**
Improved grassland (3031.67 ha)	2240.79ha (74%)	Improved grassland
	656.71ha (22%)	Cultivated/disturbed land -arable
Coniferous woodland plantation (147.09 ha)	118.71ha (81%)	Coniferous woodland plantation
	19.11ha (13%)	Mixed woodland plantation
Acid grassland - unimproved (712.77 ha)	249.25ha (35%)	Coniferous woodland plantation
	107.21ha (15%)	Acid grassland - unimproved
	72.44ha (11%)	Improved grassland
Dry dwarf shrub heath (139.49ha)	82.91ha (59%)	Dry dwarf shrub heath - acid
	22.01ha (16%)	Dry heath/acid grassland
	34.57ha (25%)	Acid grassland - unimproved
Mixed woodland plantation (273.72ha)	139.86ha (51%)	Mixed woodland plantation
	127.07ha (46%)	Coniferous woodland plantation
	34.06ha (12%)	Cultivated/disturbed land -arable
Bracken (252.09 ha)	76.18ha (30%)	Poor semi-improved grassland
	68.05ha (27%)	Bracken
	35.97ha (14%)	Acid grassland - unimproved
Built land (283.99 ha)	275.16ha (97%)	Built land
Gardens (1.63 ha)	0.44ha (27%)	Acid grassland - unimproved
	0.28ha (17%)	Broadleaved woodland plantation
	0.22ha (14%)	Improved grassland
	0.7ha (43%)	Gardens
Standing water (117.17ha)	114.49ha (98%)	Standing water
Marsh/marshy grassland (386.56 ha)	172.28ha (45%)	Acid grassland - semi-improved
	66.48ha (17%)	Coniferous woodland plantation
	59.6ha (15%)	Marsh/marshy grassland
Fen-valley mire (76.22 ha)	33.13ha (44%)	Wet heath/acid grassland
	27.25ha (36%)	Fen - valley mire
Dry heath/acid grassland (309.53 ha)	192.4ha (62%)	Dry heath/acid grassland
	37.09ha (12%)	Dry dwarf shrub heath - acid
	31.82ha (10%)	Bracken
Bogs (4163.36 ha)	2245.23ha (54%)	Coniferous woodland plantation
	911.43ha (22%)	Bogs
Cultivated land - arable (2857.99 ha)	1647.17ha (58%)	Cultivated/disturbed land -arable
	1122.19ha (39%)	Improved grassland
Poor semi-improved grassland (730.42 ha)	257.7ha (35%)	Improved grassland
	202.72ha (28%)	Cultivated/disturbed land -arable
	151.13ha (21%)	Poor semi-improved grassland
Acid grassland - semi-improved (924.81 ha)	343.43ha (37%)	Acid grassland - semi-improved
	115.79ha (13%)	Coniferous woodland plantation
	108.6ha (12%)	Acid grassland - unimproved
	94.52ha (10%)	Wet modified bog
	93.31ha (10%)	Improved grassland
Wet dwarf shrub heath (322.44 ha)	99.39ha (31%)	Coniferous woodland plantation
	89.81ha (28%)	Wet heath/acid grassland
	45.28ha (14%)	Wet dwarf shrub heath - acid
	43.6ha (14%)	Fen - valley mire
Wet heath/acid grassland (1489.42 ha)	710.59ha (48%)	Wet heath/acid grassland
	491.59ha (33%)	Acid grassland - semi-improved
Broadleaved woodland (431.60 ha)	170.93ha (40%)	Broadleaved woodland plantation
	99.06ha (23%)	Improved grassland
	71.38ha (17%)	Mixed woodland plantation
	62.28ha (14%)	Coniferous woodland plantation
Scrub (167.58 ha)	55.63ha (33%)	Coniferous woodland plantation
	40.91ha (24%)	Mixed woodland plantation
	23.83ha (14%)	Scrub
	22.12ha (13%)	Improved grassland
Neutral grassland (151.15 ha)	50.59ha (34%)	Coniferous woodland plantation
	24.2ha (16%)	Improved grassland
	22.09ha (15%)	Mixed woodland plantation
	18.54ha (12%)	Neutral grassland
	18.08ha (12%)	Cultivated/disturbed land -arable
Flush and spring - acid/neutral flush (79.81 ha)	14.01ha (18%)	Wet heath/acid grassland
	13.79ha (17%)	Coniferous woodland plantation
	13.4ha (17%)	Improved grassland
	10.61ha (13%)	Dry dwarf shrub heath
	7.91ha (20%)	Flush and spring - acid/neutral flush
	8.44ha (11%)	Standing water
Excavation site/Quarry (12.85 ha)	5.24ha (41%)	Mixed woodland plantation
	4.14ha (32%)	Neutral grassland
	2.34ha (18%)	Standing water

Tables 4-2 and 4-3 show that most semi-natural habitat types changed to intensively habitat types by 2009. Area occupied by most semi-natural habitat types such as acid grassland, marshy grassland, bogs, heath and neutral grassland in 1946 changed into coniferous woodland plantations by 2009 in both catchments. The above habitat transition matrices also show that other semi-natural habitat types such as dwarf shrub heath changed to heath/acid grassland mosaics, which could be an indication of the natural succession process.

On the other hand, intensively managed habitat types such as areas occupied by improved grassland in 1946 either remained the same by 2009 or changed into another intensively habitat type, mainly cultivated land. For example, Table 4-3 shows that 74% of the total area under improved grassland in the Ale catchment in 1946 remained under improved grassland in 2009 while 22% of area under improved grassland area in 1946 changed to cultivated land by 2009. Overall, most semi-natural habitat types changed into either improved grassland or coniferous woodland plantations by 2009 in both catchments as also illustrated by the net change habitat maps in the next section.

4.2.3 Spatial location of habitat changes within the study catchments

In order to identify the spatial location of habitat types that changed and those that remained the same between 1946 and 2009, a net change map for each of the study catchments was derived from the polygon overlay function in ArcGIS. To do this, the polygon layers (habitat maps) from the two dates and their attributes were overlaid to produce a new polygon layer showing polygons who attributes changed and those that remained the same. The net change maps (figure 4-11 and 4-13) below show the spatial location of areas where there were changes from one habitat type to another and also those areas where there was no change in habitat type in the study catchments.

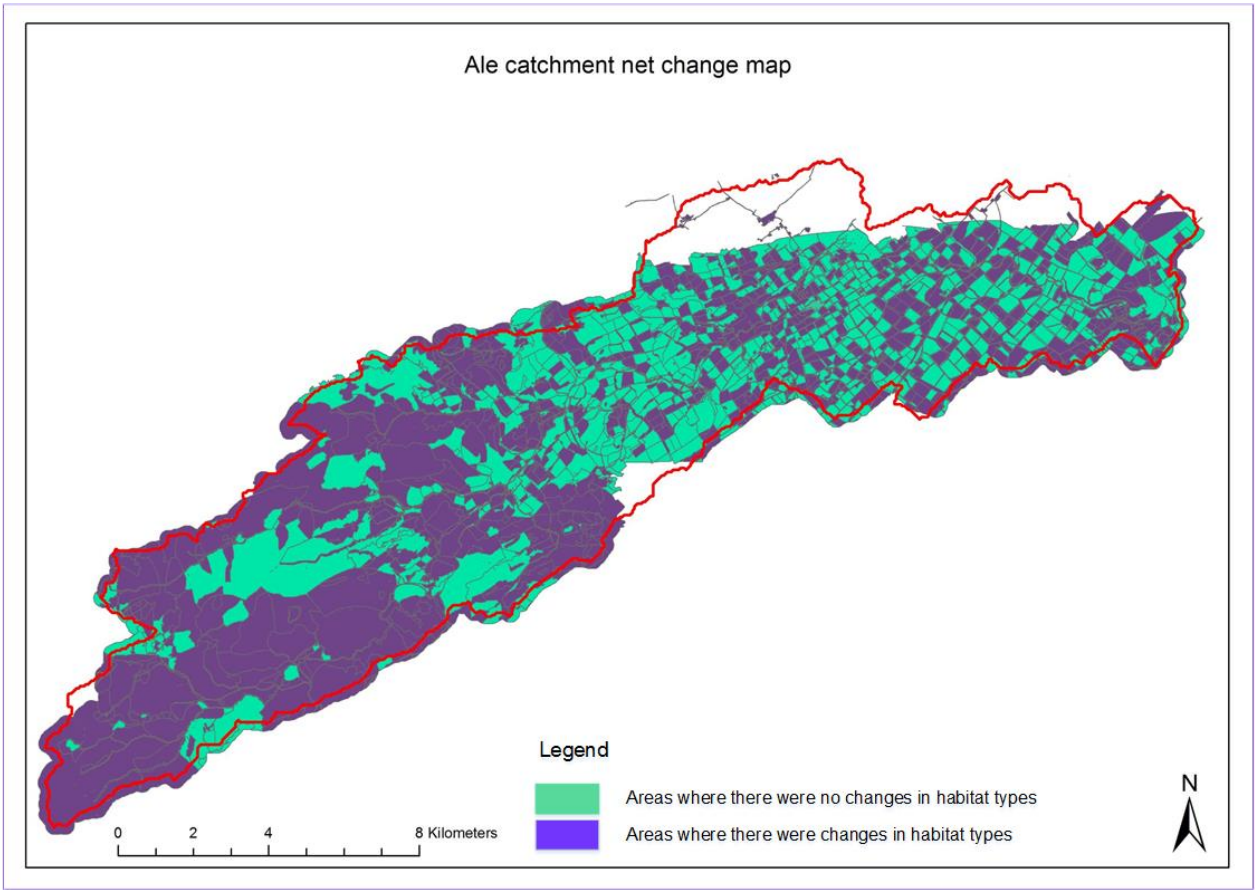

Figure 4-11: Net habitat change map for the Ale catchment

Figure 4-11, shows that most habitat type changes occurred in the upper and, to a lesser extent in the lower catchment areas of the Ale. The upland areas of this catchment are to the South West; herein referred to as the upper catchment. In 1946, the uplands were mainly dominated by semi-natural habitat types such as bogs and heath (wet dwarf and wet heath/acid grassland mosaics) and most of these areas had changed by 2009. Such changes are evident towards the top most end of the upper catchment where coniferous woodland plantations were introduced. However, there are some areas that did not change as shown in the map e.g. reservoirs and areas occupied by remaining modified bogs and other semi-natural habitats like unimproved acid grassland.

The map also shows that other habitat type changes occurred in the lower catchment of the Ale; these are the low lying areas of this catchment dominated by farming activities and settlement areas. The changes in the lower catchment mainly emanated from the reduction in cultivated land and increase in improved grassland and coniferous woodland in 2009. Other spatial changes resulted from the removal of hedgerows, neutral grassland and introduction of other habitat types such as caravan sites. However, these other changes were at a smaller scale compared to the increase of improved grassland and coniferous woodland.

Most of the area that remained unchanged was the mid catchment of the Ale. These were mainly occupied by improved grassland areas in 1946 and remained so in 2009.

Illustration on changes in spatial location of dominant habitat types in the Ale catchment

The map below is an illustration of the spatial changes in the location of the habitat types that increased to be the most dominant in the study catchments in 2009. As discussed in the preceding sections, the greater proportion of the catchment areas in 2009 was under coniferous woodland plantations and improved grassland. The maps (Figure 4-12) shows the changes to the spatial location of these most dominant habitat types between the two dates in the Ale catchment.

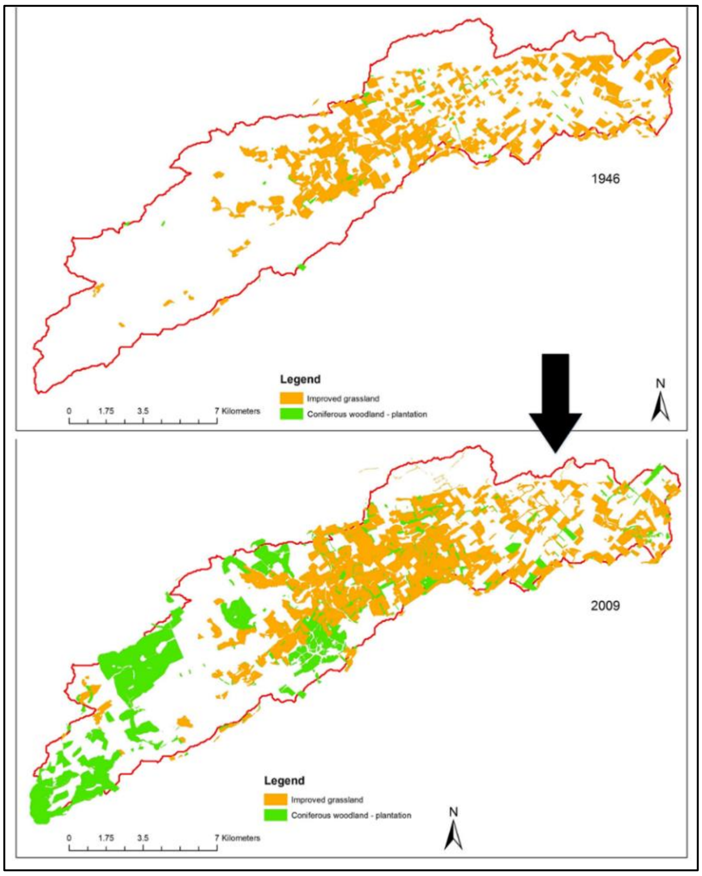

Figure 4-12: Changes to spatial location of dominant habitat types in the Ale catchment in 1946 and 2009

Figure 4-12 shows that in the Ale catchment major habitat changes observed in the upper catchment areas were mainly due to the expansion of coniferous woodland plantations; which were not there in 1946. In 1946, as shown in the top map, coniferous woodland plantations were limited in extent with few occurrences in the lower catchment, where they were mainly in linear patterns as field margins and boundaries.

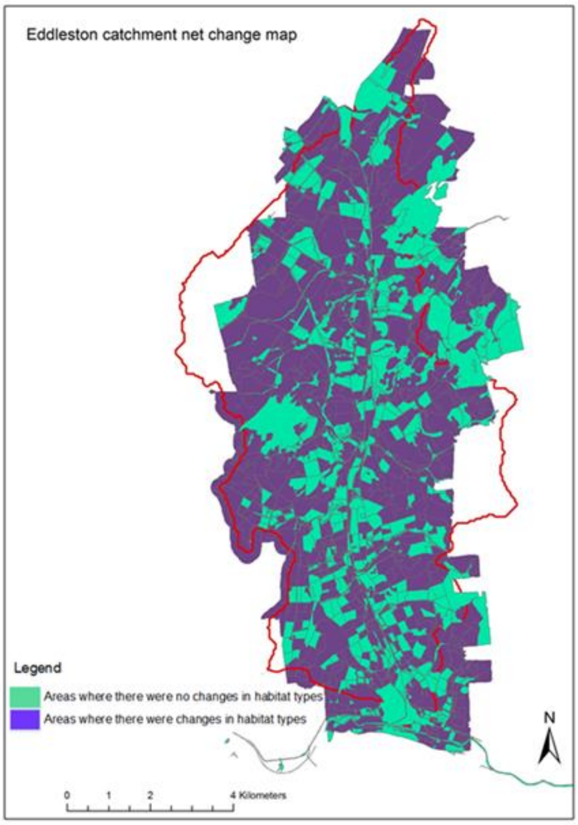

Figure 4-13: Net habitat change map for the Eddleston catchment

By 2009, this habitat type spread to occupy large expansive stands in the uplands. This change in spatial location of coniferous woodland did not only replace semi-natural habitat types such as bogs but it also led to the interspersing of other semi-natural habitats like acid grassland which in 1946 existed as continuous stands. Also in the mid and lower catchment areas of the Ale, expansive stands of coniferous woodland are noticeable in 2009 where they not only occur in linear patterns but also as extensive stands.

On the other hand, improved grassland areas in 2009 increased in concentration in the mid catchment. It also further spread to the lower catchment and towards the marginal areas of the upper catchment with few occurrences in the upper catchment areas.

In the Eddleston catchment, a similar pattern of spatial location of changes was also evident. This catchment`s uplands flank either sides of the Eddleston valley. Habitat types found in the uplands in 1946, mostly the semi-natural habitats, were spatially displaced mainly by coniferous woodland plantations by 2009. By then, where coniferous woodland plantations existed as expansive stands on the western upland areas, south east uplands and at the north eastern part of the top of this catchment.

In the Eddleston valley floor, the low lying areas of this catchment, improved grassland spread to occupy most of the area and also spread to the marginal areas towards the uplands, in some cases adjacent to coniferous woodland. Other changes within the Eddleston valley floor were in the reduction of cultivated land and poor semi-improved grassland areas.

Other areas like the settlement areas of Peebles did not change spatially. There also were areas around the Portmore Loch that remained the same between the two dates.

The net change maps for both study catchments show that there were habitat type changes in both the upland and lowland areas of these catchments between 1946 and 2009. In the uplands the changes in habitat types resulted from the introduction of intensively managed habitat types which either replaced or were interspersed with semi-natural habitat types. While this also applied to the lowland areas, these areas were already dominated by intensively managed habitat types in 1946 and the changes observed were in the expansion of one intensively managed habitat type i.e. improved grassland and the reduction of another such cultivated land.

4.2.4 Measures of landscape, habitat pattern and fragmentation

Ecological aspects such as the configuration of habitat patches within the landscape, their connectivity and diversity influence how habitats function to deliver ecosystem services. This will be discussed in detail in the next chapter but as an example, such ecological aspects influence biodiversity by supporting species population dynamics, species movement patterns, their size distribution patterns and migration rates (Fahrig, 2007). In order to understand these ecological aspects in the study catchments, landscape metrics indicating habitat pattern, fragmentation and diversity were computed. These included number of habitat patches, average habitat patch size, habitat patch size standard deviation and the Shannon diversity index. These selected landscape metrics are noted to be good indicators of habitat fragmentation (Soons et al., 2005, Kienast, 1993, Antwi et al., 2008, Johansson et al., 2008, Fahrig, 2003).

FRAGSTATS (version 4.1) was used to compute these measures with outputs from these computations including statistics and indices at habitat type level and entire catchment (landscape) levels. Habitat type level indices measure the configuration of habitat types within the landscape, while the landscape level indices measure the overall landscape pattern (McGarigal et al., 2012). Table 4-2 below presents the results of landscape/catchment level metrics, while appendix 15a and b provide the habitat type level metrics for the study catchments. (Refer to appendix 14 for a detailed description of all the metrics shown in Table 4-2).

Table 4-4: Landscape level metrics for the Ale and Eddleston catchments

Landscape Metrics	Eddleston catchment			Ale catchment		
	1946		2009	1946		2009
Patch density and size metrics						
Number of patches (NUMP)	1393	↑	3132	2166	↑	6789
Mean patch size (MPS) (ha)	5.78	↓	2.55	7.88	↓	2.51
Patch size standard deviation (PSSD)	10.34	↓	7.97	24.61	↓	14.24
Patch size coefficient of variance (PSCV)	178.89	↑	312.55	312.31	↑	567.33
Proximity Index						
Mean Proximity Index (MPI)	68.22	↑	197.25	171.99	↑	207.4
Edge metrics						
Edge density (ED) (m/ha)	120.92	↓	103.33	108.44	↓	101.28
Shape metrics						
Mean shape Index (MSI)	1.67	↓	1.58	1.98	↓	1.71
Diversity metrics						
Shannon`s Diversity Index (SDI)	2.76	↓	2.23	2.74	↓	2.48

4.2.4.1 Indicators of habitat fragmentation and connectivity

Number of habitat patches

As shown in Table 4-2, the number of habitat patches[15] in both catchments increased between the two time periods. In the Ale catchment, the number of patches increased by more than three times from approximately 2166 in 1946 to about 6789 in 2009. Similarly in the Eddleston, the patch numbers doubled from 1393 in 1946 to 3132 in 2009. The increase in the number of habitat patches was much bigger in the Ale catchment than the Eddleston. The increase in number of habitat patches in 2009 reveals fragmentation of habitats into smaller sizes over this period. Though these results show that there was an overall increase in the number of habitat patches at the catchment level, there were, however, variations among different habitat types as shown in appendix 15a and b. These were, for example the habitat types that recorded an overall decrease in their extent in 2009, such as broadleaved woodland or arable land in the Eddleston catchment.

Mean patch sizes

Overall, in both catchments, the mean patch sizes decreased to more than half the mean patch sizes in the 1940s (Table 4-2). The average patch size reduced by nearly 50% from 5.78 ha in 1946 to 2.55 ha in 2009 in the Eddleston while it reduced by over 60% in the Ale catchment. This further indicates that the 2009 catchment landscapes are more fragmented compared to the 1940s.

However, the mean patch sizes varied among the different habitat types within the study catchments (Appendix 15a and b). The most significant reduction in the mean patch sizes was for wet modified bogs and wet heath/acid grassland mosaics. This could imply that semi-natural habitats such as these were more fragmented as they increasingly became interspersed with other habitat types like coniferous woodland plantations. A reduction in mean habitat patch size was, however, not apparent for habitat types such as coniferous woodland plantation and enclosed habitat types like improved grassland, as some of these instead increased in areal extent.

[15] A habitat patch is a discrete cluster of pixels or a polygon representing a habitat type found in the study catchments. The minimum area of a habitat patch in this study equates to the minimum mapping unit used in this study i.e. 0.1 ha (refer to chapter 3).

Proximity Index

The proximity Index measures the degree of isolation between habitat patches of the same type within a set search radius. A high proximity index is an indication of habitat patches of the same type are located close to each other. Table 4-2 shows that the mean proximity index[16] was higher in 2009 in both catchments. This is likely to have been the influence of habitat types like improved grassland and coniferous woodland plantations which increased to be the most dominant habitat types in the study catchments as they accounted for greater proportions of these catchment areas in 2009.

Analysis of the proximity index at the habitat type level showed that this varied among different habitat types (refer to appendix 15a and b). In 1946, the proximity index was higher for semi-natural habitat types such as wet modified bogs and heath/acid grassland mosaics but it decreased in 2009 as these habitat types became interspersed with coniferous woodland plantations and in the process fragmented into smaller sizes. Conversely, in 2009, the proximity index was higher among intensively managed/modified habitat types such as improved grassland though in the Eddleston catchment, the proximity index for cultivated/disturbed land also reduced due to the overall reduction in the presence of this habitat type in 2009.

4.2.4.2 Indicators of habitat diversity and heterogeneity

Patch size standard deviation

As shown in the table, both catchments had a higher patch size standard deviation (PSSD) in 1946 than in 2009. In the Ale catchment the patch size standard deviation was greater in 1946 (24.61) and it reduced by nearly 50% to approximately 14.24 in 2009. Similarly, in the Eddleston catchment, the PSSD reduced by about 67% from 10.34 in 1946 to 7.97 in 2009. This could be suggestive of homogenous catchment landscapes in 2009 compared to the past as a high PSSD indicates a more heterogeneous landscape. Such landscape uniformity in these catchments is also reflected in the 2009 habitat maps (Appendix 10a and b), which show a rather greener, more wooded, less diverse landscape compared to the 1946 habitat maps.

[16] Estimated using a search radius of 500m as an average distance for functional connectivity for generalist woodland species (http://www.forestry.gov.uk/fr/infd-69pla5#gfs)

Edge density

Edge density measures the amount of border present between patches of different habitat types, indicating spatial heterogeneity within the landscape. A higher edge density is a reflection of a high degree of spatial heterogeneity (McGarigal et al., 2012). In both catchments, the edge density decreased between the two time periods; further implicating a less diverse landscape in 2009 compared to 1946. Between the two catchments, the edge density decreased much more in the Eddleston catchment than in the Ale catchment in 2009, though the Eddleston had a slightly higher edge density than the Ale in 1946. This could imply that this catchment lost more diversity than the Ale.

Shannon diversity Index

The Shannon diversity Index quantifies the diversity of a landscape based on the number of different habitat patch types and the proportional area distribution among the patch types. It increases as the number of different habitat patch types increases; indicating more landscape diversity (Magurran, 2004).

Results from this study show that the Shannon diversity Index decreased in 2009 in both catchments. The Eddleston catchment again recorded a lower diversity index compared to the Ale catchment in 2009 though it had a slightly higher index than the Ale catchment in 1946. This could be due to the dominance of improved grassland in the Eddleston as it increased to account for nearly 43% of the total catchment area in 2009, compounded by its small size compared to the Ale catchment; which is twice as big. Such a reduction further confirms a shift from more diverse catchment landscapes in 1946 to less diverse ones in 2009 with the Eddleston being much less diverse than the Ale catchment.

4.2.5 Section summary

Spatial changes to habitats in the study catchments have been assessed from three aspects i.e. (a) changes in their extent (area) between 1946 and 2009, (b) changes to their spatial location and (c) patterns of spatial change and the configuration of habitats within the wider catchment landscape. Findings show that both catchments depicted broadly similar habitat change trend between 1946 and 2009.

Such similar trends included a significant increase in intensively managed habitat types, notably improved grassland and coniferous woodland in both catchments as these increased to account for greater catchment proportions in 2009. Coniferous woodland was less common in 1946 and it recorded the highest percentage increase in 2009 compared to other habitat types in both catchments. Improved grassland was already dominant in 1946 in both catchments and further increased to remain so in 2009. The Eddleston catchment registered a much higher percentage increase in improved grassland compared to the Ale catchment.

On the other hand, area under semi-natural habitat types like bogs had massively decreased by 2009 in both catchments. These recorded greater percentage decreases compared to other habitat types. The Eddleston catchment also recorded a significant reduction of about 80% in area under arable land in 2009 compared to the Ale catchment which recorded a 13% decrease. However, there were also other habitat types that increased in one catchment and decreased in the other such as marshy grassland.

Spatial changes in the location of habitats were also similar in both catchments. The upland areas of the study catchments recorded significant changes following the introduction of coniferous woodland. Semi-natural habitat types in these areas like bogs, wet dwarf shrub heath and acid grassland were either replaced or interspersed with coniferous woodland and other habitat types like bracken. Spatial changes in the low lying areas of these catchments were related to changes from one intensively managed habitat type to the other especially the increase of improved grassland as well as the removal of hedgerows.

Consequently, the introduction and dominance of intensively managed habitat types by 2009 resulted in more fragmented and less diverse catchment landscapes as evidenced by the increase in the number of habitat patches, low Shannon Diversity Index etc. The Eddleston catchment was the least diverse between the two catchments by 2009 despite almost equal diversity indices in 1946. This raises the question as to whether the Eddleston catchment changed the most compared to the Ale or it is merely the influence of its small size compared to the Ale catchment.

4.3 Section 2: Changes in Ecosystem services delivery between 1946 and 2009

This section presents the results from analysis of changes in ecosystem services delivery in the study catchments, guided by the following research questions:

c. What are the historic and current ecosystem services supplied in the Ale and Eddleston catchments?
d. What spatial changes have occurred to these between the 1940s and the early 21st century?
e. How have the identified changes in habitats influenced changes in ecosystem services delivery in the study catchments?

In response to these research questions, an analysis of spatial changes in the location, extent of ecosystem service supply areas and relative levels of ecosystem service supply in the study catchments in 1946 and 2009 was done. The ecosystem services maps presented here show the spatial location of ecosystem service supply areas between the two dates in the study catchments. Zonal statistical analysis (frequency counts) were done on the ecosystem services maps (raster format) produced in stage 3 of the data collection and processing stage (refer to chapter 3) to identify the extent[17] (ha) of ecosystem service supply areas in 1946 and 2009 (appendix 17). Changes to the extent of these supply areas were assessed in relation to qualitative ecosystem service supply levels of low, medium and high. These relative levels of ecosystem service supply are in line with the SENSE method ecosystem services assessment approach, explained in the previous chapter. Bar graphs summarising percentage of total catchment areas and their ES supply levels between the two dates are aligned to the respective ecosystem service maps.

As explained in the previous chapter, ecosystem services for crop, timber and livestock production were assessed based on changes in area under arable land, coniferous woodland and improved grassland respectively. For this reason, zonal frequency counts and relative levels of low, medium, high were not done for these ecosystem services. Instead the area changes for these were coupled with national level estimates on changes in yields for these ecosystem services between the 1940s and 2009.

[17]The frequency counts were divided by the number of pixels making up a hectare (100 in this case) in order to estimate catchment areas (ha) supplying the different relative ES levels i.e. medium, high etc. between the two dates.

To present changes in ecosystem service delivery, ecosystem service maps per ecosystem service type for the two dates are described under the relevant sub headings. Such changes in ecosystem services are also linked to associated habitat changes presented in the previous section.

4.3.1 Ecosystem services for which supply capacity increased between 1946 and 2009

Four ecosystem services showed an increase in supply capacity between 1946 and 2009. These were flood control, vegetation carbon storage, timber provision and livestock production. They also increased in spatial extent to account for greater catchment areas. These are in turn discussed below.

Flood control (regulating ES)

Flood control refers to the water retention and storage function of habitats in delaying the release of water from land surface to water courses while also promoting water infiltration to protect against flooding. Other influencing factors include the type of underlying soil, geology and topography.

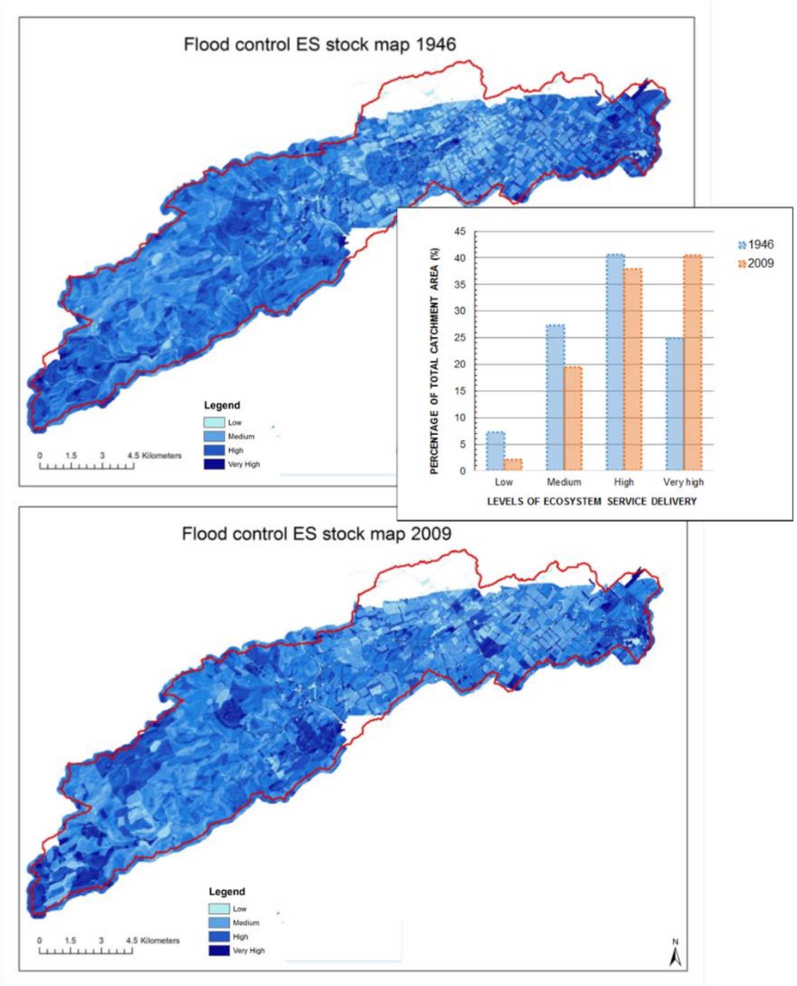

Figure 4-14: Areas important for flood control in the Ale catchment

The flood control ecosystem service maps (figures 4-14 and 4-15) show that in both time periods, the uplands of the study catchments had a high capacity to control flooding. In 1946, these catchments had a relatively medium-high capacity to control flooding. Such high flood control areas were associated with semi-natural habitats and broadleaved and mixed woodland which were present in the catchments at that time.

By 2009, there was an increase in high-very high flood control areas as indicated by the darker colour intensities on the 2009 maps, some of which spread to the low lying areas of these catchments. The capacity to control flooding further increased in 2009 when more woodland plantations were introduced especially in the upland areas which are prone to increased runoff if vegetation cover is limited. Woodland has a high capacity to regulate runoff and promote water infiltration in such areas.

In addition, the reduction in arable land in the low lying areas of these catchments especially in the Eddleston catchment also improved the flood control capacity as land cultivation promotes runoff. The maps show that the Eddleston catchment had a much higher potential for flood regulation compared to the Ale in both time periods. This could be linked to the extent and shape of this catchment compared to the Ale which is expansive and has three distinct sections with varying topography, vegetation cover and soil types which influence flooding.

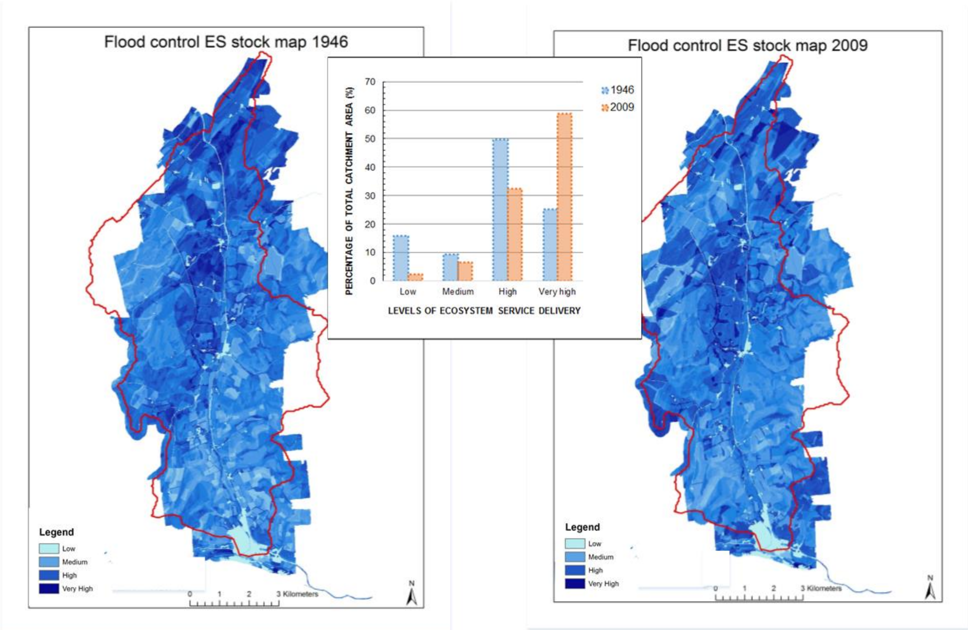

Figure 4-15: Areas important for flood control in the Eddleston catchment

Bar graphs for this ecosystem service show an increase in the supply levels of this ecosystem service by 2009 in both catchments. In 1946, about 40% of the Ale total catchment area had a high capacity to regulate flooding (Figure 4-14) while about 50% of the Eddleston catchment (Figure 4-15) area had a high capacity for flood control. Only a few places within the lower catchment areas of the Ale and Eddleston valley floor had a low capacity to control flooding. These were mainly areas associated with built land and arable land.

By 2009, there had been a shift to very high flood control capacities in the study catchments. In the Ale catchment, about 40% of the total catchment area had a very high capacity to regulate flooding. There were, however, areas with high capacity and these accounted for about 35% of the total catchment area. In the Eddleston about 60% of the total catchment area had a very high capacity to regulate flooding during the same period.

Vegetation carbon storage (regulating ES)

Vegetation carbon storage refers to the sequestration and storage of atmospheric carbon by vegetation.

Both catchments show a general increase in vegetation carbon storage capacity in 2009. In particular, there was an increase in very high vegetation carbon storage areas in the uplands of these catchments. The maps for flood control and vegetation carbon show spatial overlaps in the location of these high supply areas. Together with the increase in flood control capacity, the increase in this ecosystem service was also associated with increased woodland plantations in the catchments in 2009 as this habitat type increased vegetation cover in the study catchments.

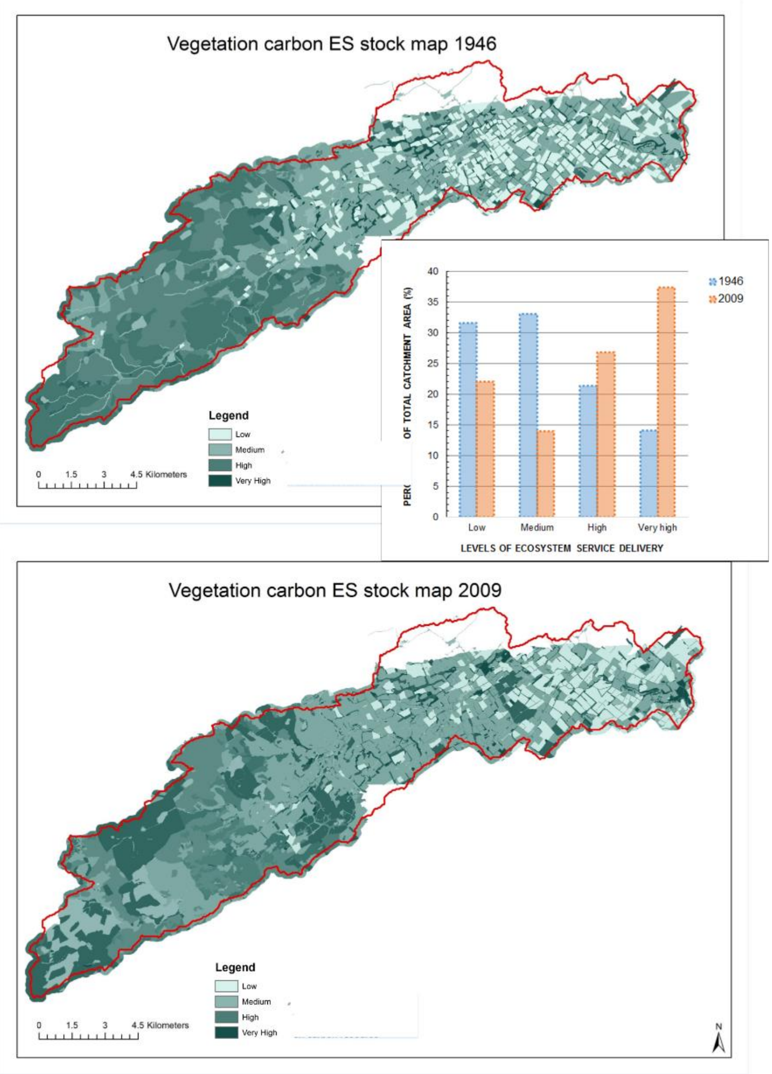

Figure 4-16: Vegetation carbon storage capacity in the Ale catchment

In 1946, the uplands of these catchments, which were dominated by bogs, heathland, acid grassland mosaics had a medium capacity to store vegetation carbon. In the low lying areas of these catchments, high vegetation carbon storage hotspots[18] were associated with broadleaved and mixed woodland plantations and hedgerows. By 2009, there was an expansion in the spatial location of high vegetation carbon storage areas to include the uplands, mainly in the upper catchment areas of the Ale.

The bar graph for the Ale catchment shows that the capacity for vegetation carbon storage increased from medium supply levels in 1946 to very high levels by 2009. In 1946, about 33% of the total catchment area had a medium capacity to supply this ES. By 2009 there was a shift to very high supply levels and such areas accounted for about 37% of the total catchment area.

In 1946, the Eddleston catchment (Figure 4-17) had high capacity vegetation carbon supply areas, accounting for about 30% of the total catchment area. At the same time, about 25% of this catchment`s area had a low vegetation carbon storage potential within the Eddleston valley floor, mainly associated with arable farming areas which were dominant at that time. By 2009, the capacity for vegetation carbon supply had increased to very high levels and such areas accounted for about 40% of the total catchment area.

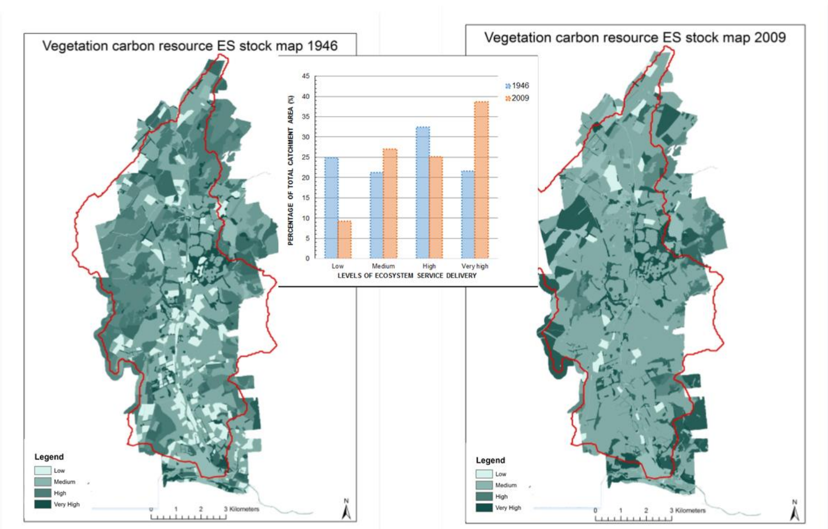

Figure 4-17: Vegetation carbon storage capacity in the Eddleston catchment

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[18] In biodiversity conservation, the term hotspot; proposed by Norman Myers in the 1980s, refers to areas of high species richness, endemism and/or threat and has been used to prioritise such areas. In this thesis the term hotspots refers to areas with high ecosystem service supply capacity indicated by darker colour intensities on the maps.

Timber production (provisioning ecosystem service)

Timber production refers to tree harvests from woodland plantations which are processed to various products such as furniture. As already mentioned, changes in area under coniferous woodland between the two dates in the study catchments was used as an indicator of their capacity to supply this ecosystem service. Coniferous woodland was selected to be an indicator for timber provision as it is the common commercial woodland in the study catchments and in the UK in general.

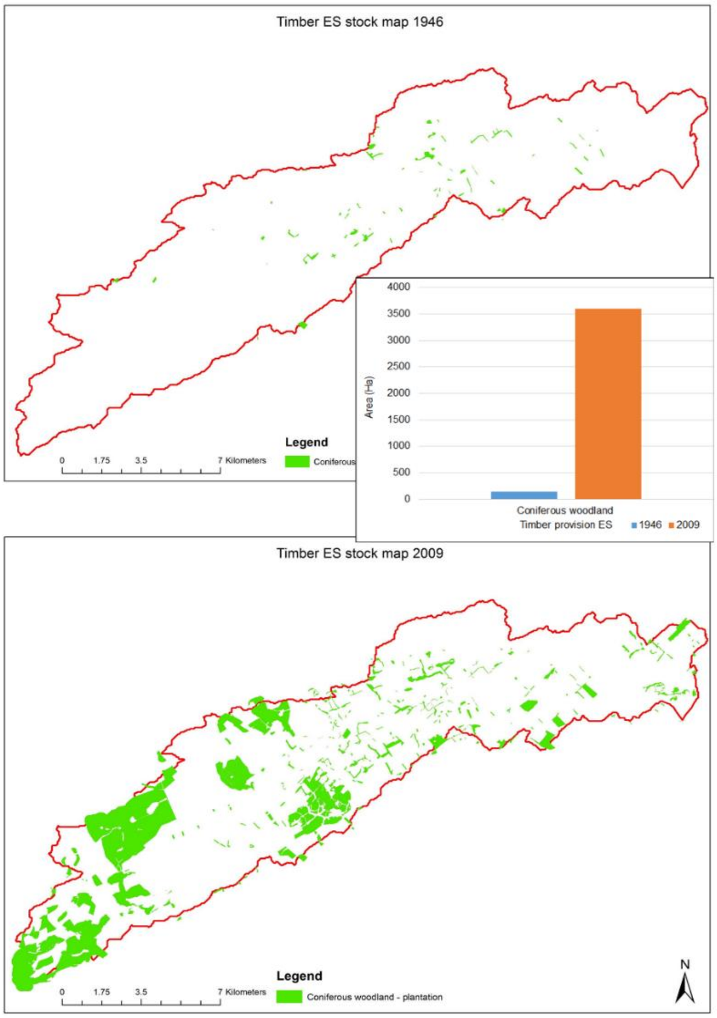

Figure 4-18: Timber provision areas in the Ale catchment

The maps (figures 4-18 and 4-19) show an increase in spatial location of timber provision areas by 2009, especially in the uplands of both catchments compared to 1946.

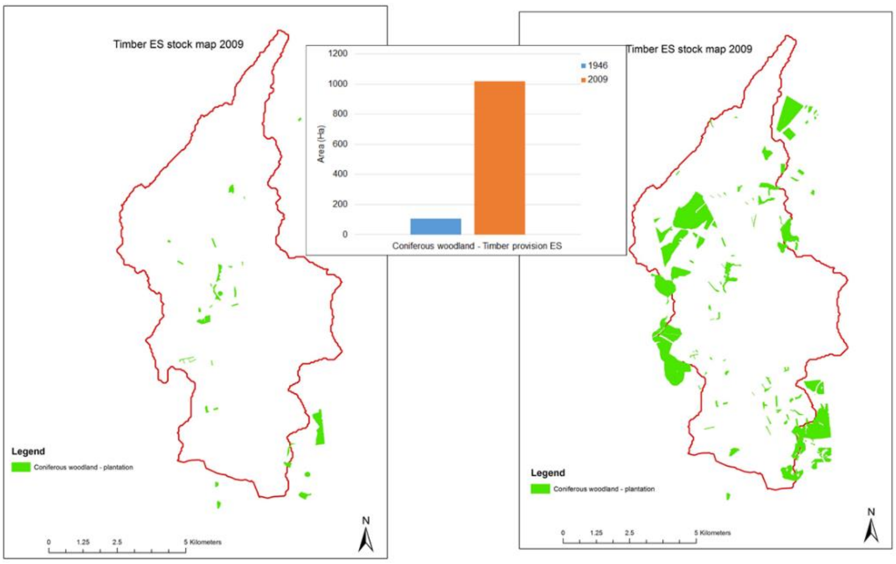

Figure 4-19: Timber provision areas in the Eddleston catchment

The bar graphs show coniferous woodland plantation areas in the study catchments in 1946 and in 2009. Coniferous woodland occupied less than 1% of the total Ale catchment area in 1946 and less than 1.4% of the total Eddleston catchment area during the same period. By 2009, coniferous woodland increased to occupy over 3000 ha (21%) of the total Ale catchment area and over a 1000 ha (13%) of the Eddleston total catchment area. The bar charts further illustrate that the greater proportion of the catchment areas were under coniferous woodland in 2009; reflecting an increased potential for timber provision.

Forestry Commission estimates show that the amount of timber harvested in Scotland since 1976 increased from 0.93 million tonnes to 5.32 million tonnes in 2008 (Forestry Commission, 2015b). This increase could provide indications on increased timber provision in the study catchments, given the presence of Forestry Commission owned land in these catchments. Furthermore, areas of recently felled coniferous woodland plantations also increased and were evident on the 2009 habitat maps (appendix 10a and b), indicating timber harvesting activities in the study catchments.

Livestock production (provisioning ecosystem service)

Livestock production refers to animals raised for domestic or commercial consumption or use, such as cattle, sheep and pigs. Changes in area under semi-improved and improved grassland were used as an indicator for livestock production as these show the extent of livestock grazing areas and source of grass harvested as livestock feed.

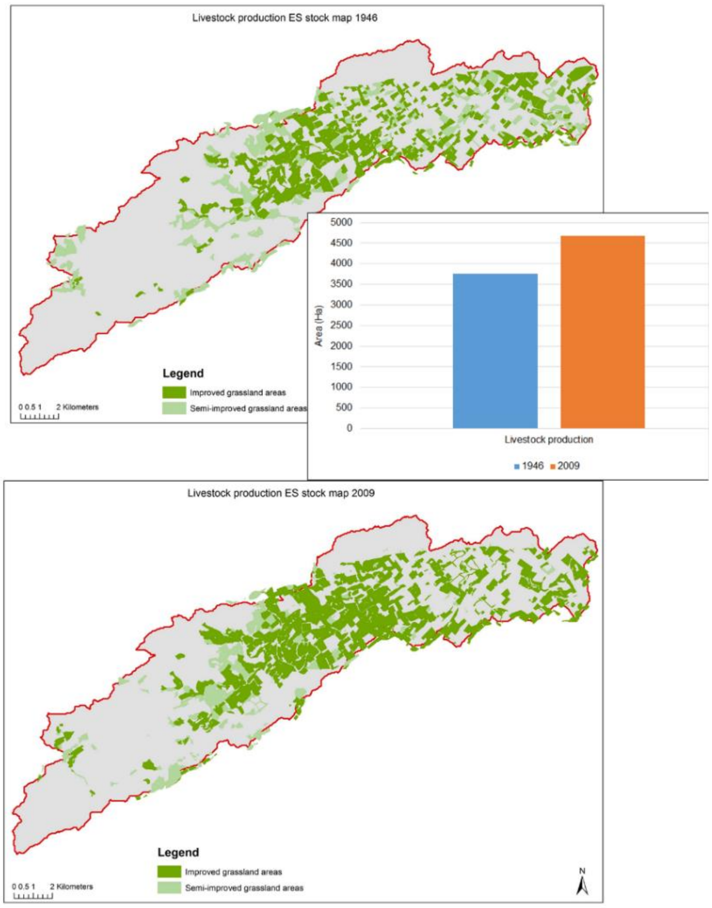

Figure 4-20: Livestock production areas in the Ale catchment

The maps (figure 4-20 and 4-21) show semi-improved and improved grassland areas in 1946 and 2009 in the Ale and Eddleston catchments. The 2009 maps for both catchments show an increase in the spatial distribution and intensity of improved grassland areas, mainly in the low lying areas of these catchments where farming activities are common. The bar graphs derived from

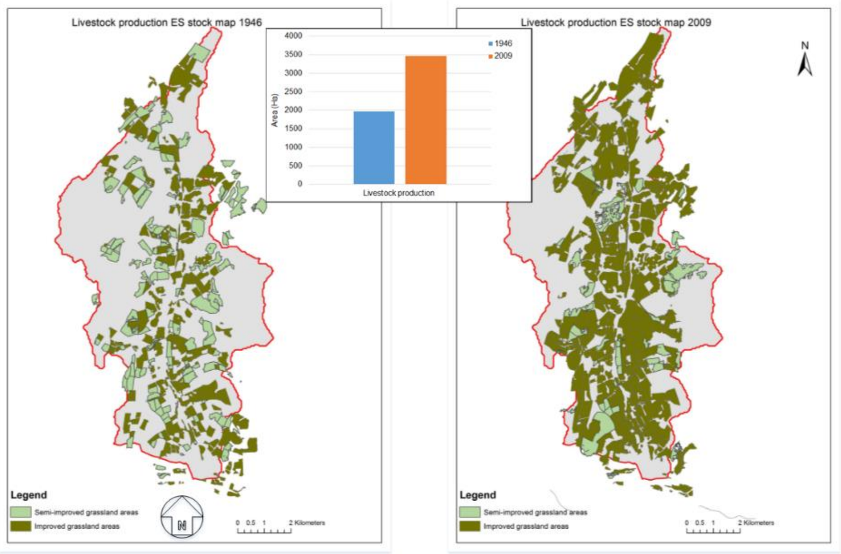

Figure 4-21: Livestock production areas in the Eddleston catchment

these maps show that a greater proportion of catchment areas was under improved grassland in 2009 compared to 1946.

Such an increase in improved grassland areas can be associated with increase in livestock numbers, livestock density and grass yields. The Scottish government agriculture census reports, estimate that sheep numbers - a major farming activity in the Scottish borders where the study catchment are located, increased from about 6 million in 1947 to almost 10 million in the 1990s (Scottish Government, 2015).

4.3.2 Ecosystem services for which supply capacity decreased between 1946 and 2009

Five ecosystem services showed a decrease in supply between 1946 and 2009. These were soil carbon storage, biodiversity, pollination resource, water quality regulation and soil erosion risk. They also decreased in spatial extent to account for less catchment areas. These are in turn discussed below.

Soil carbon storage (regulating ecosystem service)

Soil carbon storage refers to the accumulation of organic matter in soils. Such organic matter emanates from the decomposition of humus, leaf litter, growth and death of plant roots, foliage etc. Peatland (mires and blanket) bogs, semi-natural grasslands and heathland are important in soil carbon storage.

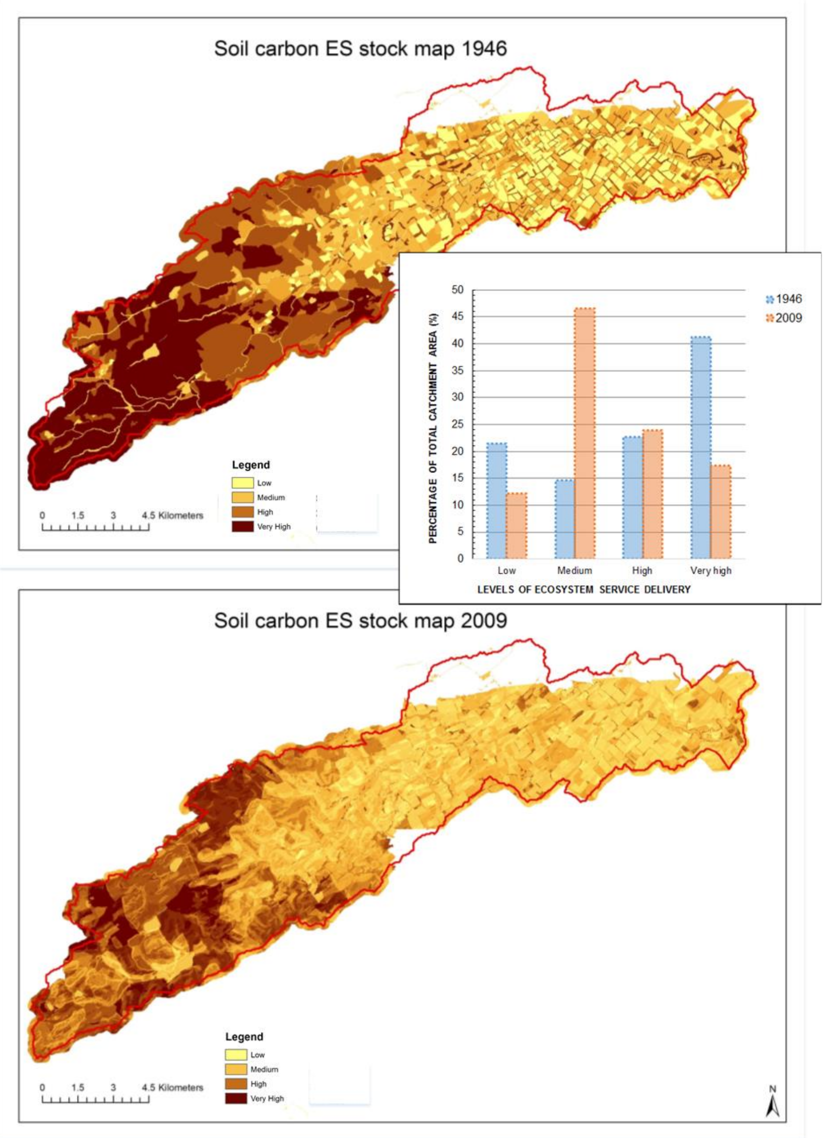

Figure 4-22: Soil carbon storage potential in the Ale catchment

The soil carbon storage ecosystem service maps (figures 4-22 and 4-23) show that the upland areas of the study catchments had a very high soil carbon storage capacity in 1946. These very high soil carbon storage areas were associated with the occurrence and spatial distribution of habitat types such as bogs (wet bogs, blanket bogs) and heath/acid grassland mosaics and unimproved acid grassland areas. Low supply areas in the low lying areas of these catchments were associated with areas under arable land as these have a low capacity to store soil carbon.

By 2009, the capacity to supply this ES had been reduced in these catchments, as area under upland semi-natural habitats like bogs decreased. However, there were a few hotspots that remained by 2009 which had very high soil carbon storage capacity, notably in the upper Ale catchment. Such hotspots showed spatial overlaps with remaining semi-natural habitat patches in the uplands. In the low lying areas of these catchments, the reduction of arable areas and increase in improved grassland by 2009 elevated the soil carbon storage capacity as grassland has a higher capacity to store soil carbon than arable land.

The bar graphs show that in the Ale catchment in 1946, large expansive areas in the upper catchment, accounting for about 40% of the total catchment area had a very high capacity for soil carbon storage. Other hotspots were also evident, occupying small areas within the lower catchment associated with woodland plantations. Likewise, in the Eddleston catchment the greater proportion of the catchment`s uplands and top of the catchment areas (40%) had a very high capacity for soil carbon storage in 1946.

By 2009, the potential to supply this ecosystem service had been reduced and there was a shift to medium soil carbon storage capacity areas (accounting for about 45% of the total Ale catchment area and 35% of the Eddleston total catchment area).

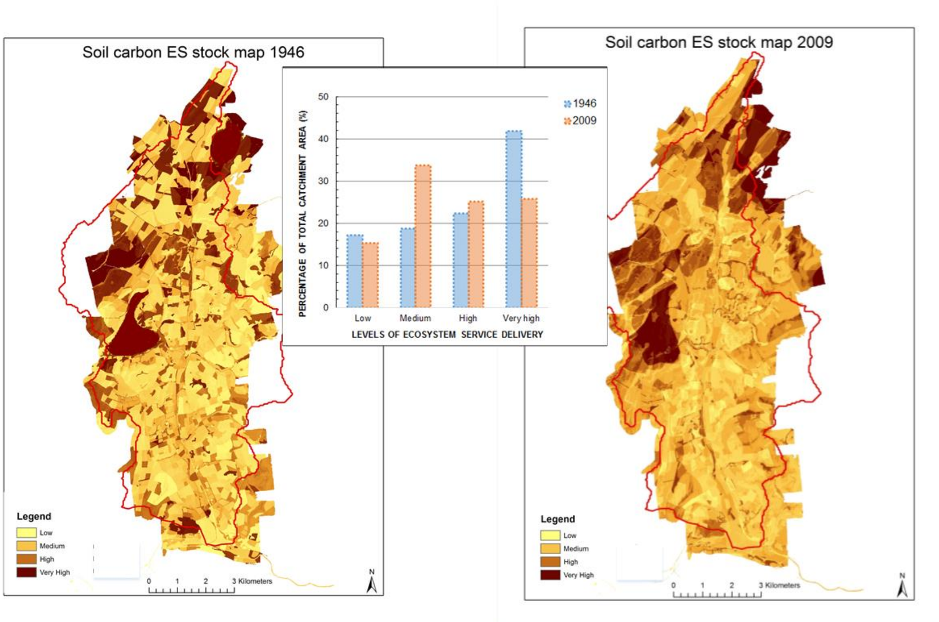

Figure 4-23: Soil carbon storage potential in the Eddleston catchment

Figure 4-23, shows a shift from very high soil carbon storage capacity in the Eddleston in 1946 to medium-high supply capacities by 2009. The soil carbon storage hotspots that were present within the Eddleston valley floor in 1946 were not evident by 2009.

Biodiversity (supporting ecosystem service)

Biodiversity was assessed based on the presence and diversity of semi-natural habitats in the study catchments between the two time periods as these have the capacity to support more diverse species of flora and fauna. Also as a supporting ecosystem service, biodiversity promotes the production of other ecosystem services.

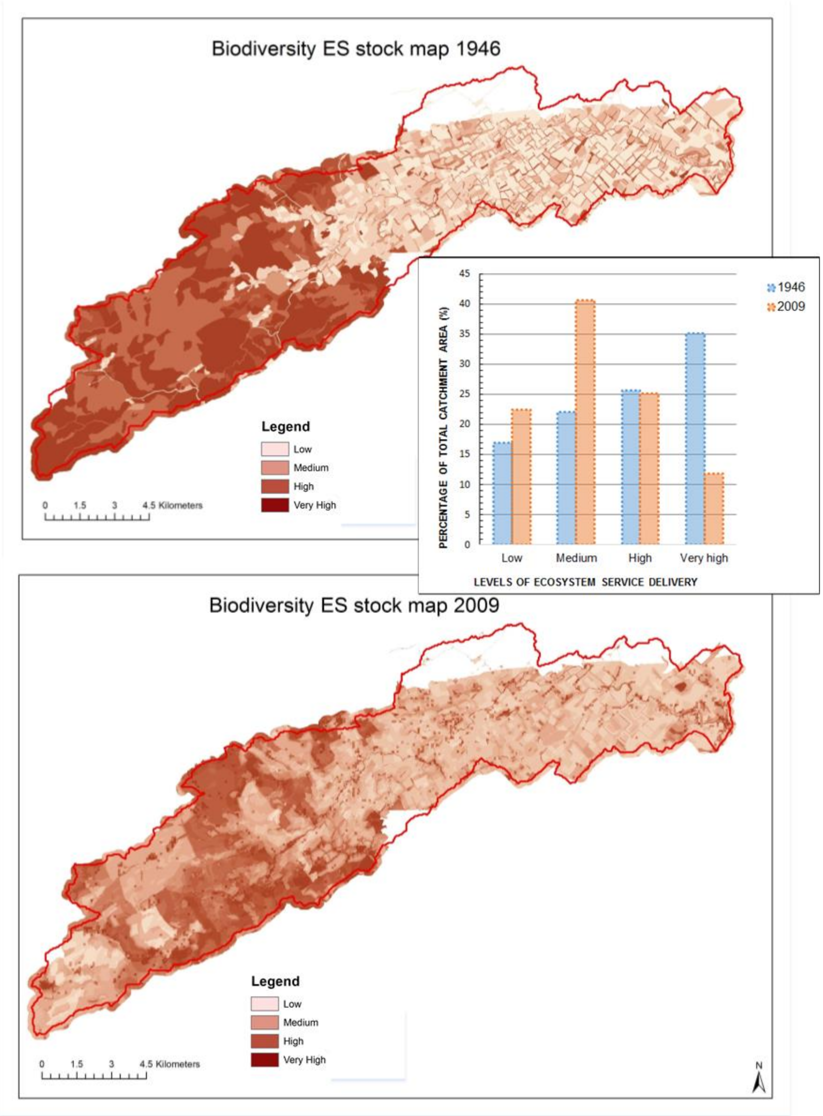

Figure 4-24: Biodiversity supply potential in the Ale catchment

The maps (figures 4-24 and 4-25) show that in 1946 the greater proportion of catchment areas, especially the uplands had more biodiverse habitats in 1946 compared to 2009, and hence a very high capacity to supply this ecosystem service.

In 1946, the uplands of these catchments had a very high biodiversity supply capacity. Typical habitat types included semi-natural habitats such as bogs, unimproved grasslands, shrub heath and heath/acid grassland mosaics. The low lying areas had low-very low capacities in 1946 except the hotspots associated with hedgerows and scrub habitat types.

The reduction and fragmentation of these habitat types by 2009 also suggests reduced capacities for the delivery of this ecosystem service in both catchments. The most marked decrease was noted in those catchment areas that were taken over by coniferous woodland plantations, as this woodland type is noted to be less biodiverse. Consequently, a shift from very high biodiversity supply areas to lower capacities ranging from medium to very low was realised by 2009. Areas with high capacities for the delivery of this ecosystem service were associated with remaining modified bogs, shrub heath and heath/acid grassland mosaics; habitats albeit interspersed with less biodiverse habitat types. Low supply capacities were recorded in the catchment areas that were dominated by intensively managed habitat types, like improved grassland e.g. in lower catchment areas of the Ale and within the Eddleston valley floor, as these are noted to be less biodiverse. Such a reduction in the biodiversity supply capacity is also supported by the decrease in the Shannon Diversity Index, discussed in section 1 of this chapter.

Bar graphs show that in the Ale catchment very high biodiverse habitats, mainly concentrated in the upper catchment in 1946, accounted for about 35% of the total catchment area. By 2009 there had been a reduction to lower ecosystem service supply levels.

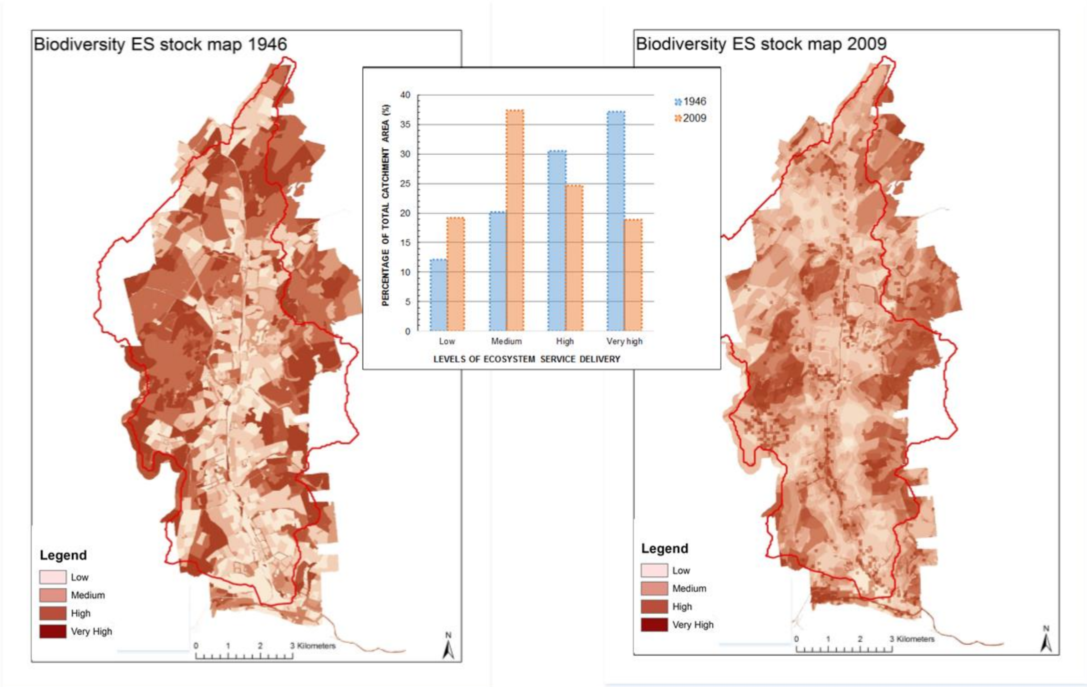

Figure 4-25: Biodiversity resource supply potential in the Eddleston catchment

In the Eddleston, the greater proportion of about 40% of the total catchment area had a very high biodiversity capacity in 1946. By 2009, there had been a shift to lower capacities and a greater proportion of the total catchment area (40%) had a medium-low capacity to supply this ecosystem service.

Pollination resource (regulating ecosystem service)

Pollination resource refers to the role of habitats in promoting the presence and spread of insect pollinators such as bees.

In both catchments, the capacity for the provision of the pollination resource ecosystem service had been reduced by 2009. Very high concentration regions for the supply of the pollination resource in 1946 were related to heathland, wet heath/acid grassland mosaics, scrub and hedgerows habitat occurrences. The medium supply areas were associated with areas occupied by bogs and unimproved acid grassland.

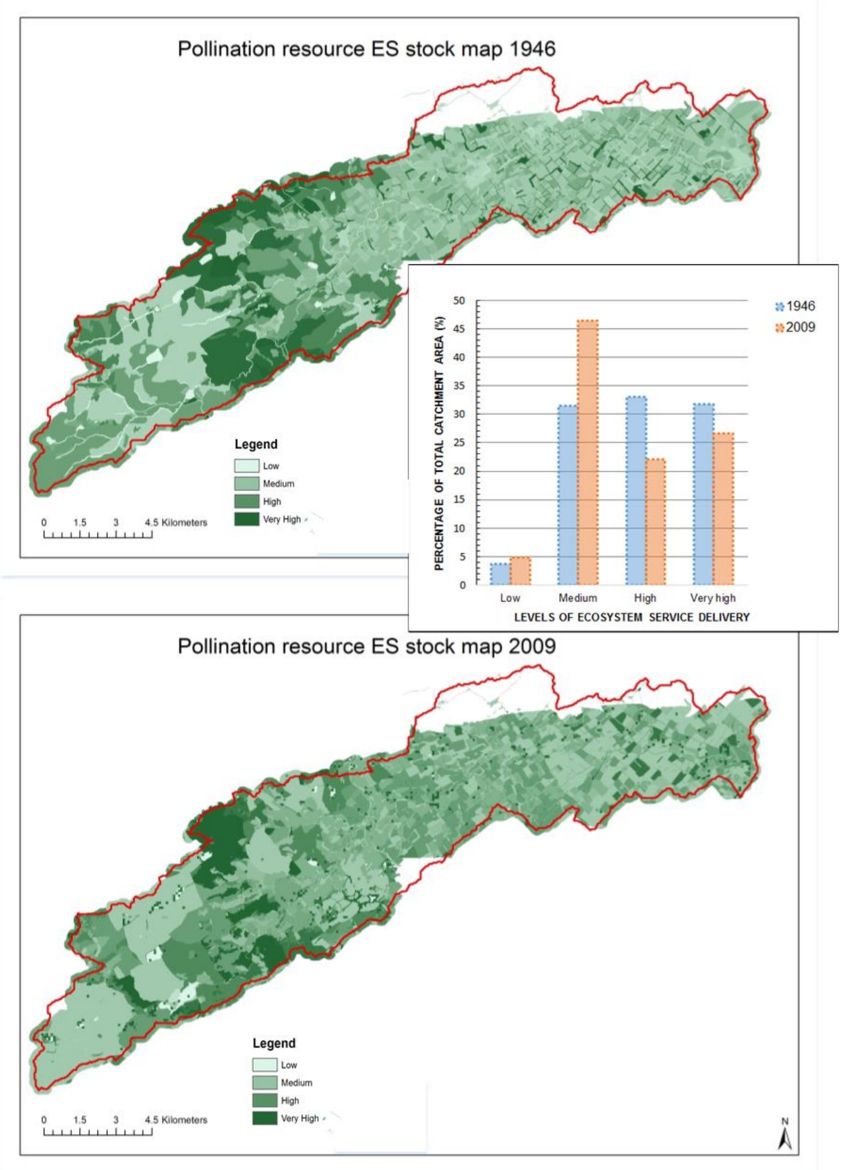

Figure 4-26: Pollination resource supply potential in the Ale catchment

Reductions in the supply areas of this ecosystem service by 2009 was noticeable in the uplands of these catchments, where coniferous woodland plantations were introduced and where these habitats with high pollination resource were removed such as, hedgerows in the low lying areas. On the other hand, the reduction in arable land areas and increase in improved grassland in the low lying areas of the catchments slightly elevated the capacity to supply this ecosystem service to medium levels by 2009. These spatial changes in the pollination resource supply areas overlap with areas where biodiversity was also reduced, reflecting the role of biodiversity in supporting such ecosystem services.

In the Ale catchment the capacity to provide this ecosystem service ranged from high to very high in 1946 (figure 4-26). Bar graphs show that, areas with high pollination resource supply levels accounted for about 33% of the total Ale catchment area while very high capacity zones accounted for nearly 32% of the Ale catchment area. By 2009, there had been a reduction in areas with a high to very high capacity to deliver this ES and the greater proportion of the total Ale catchment area (46%) had a medium capacity.

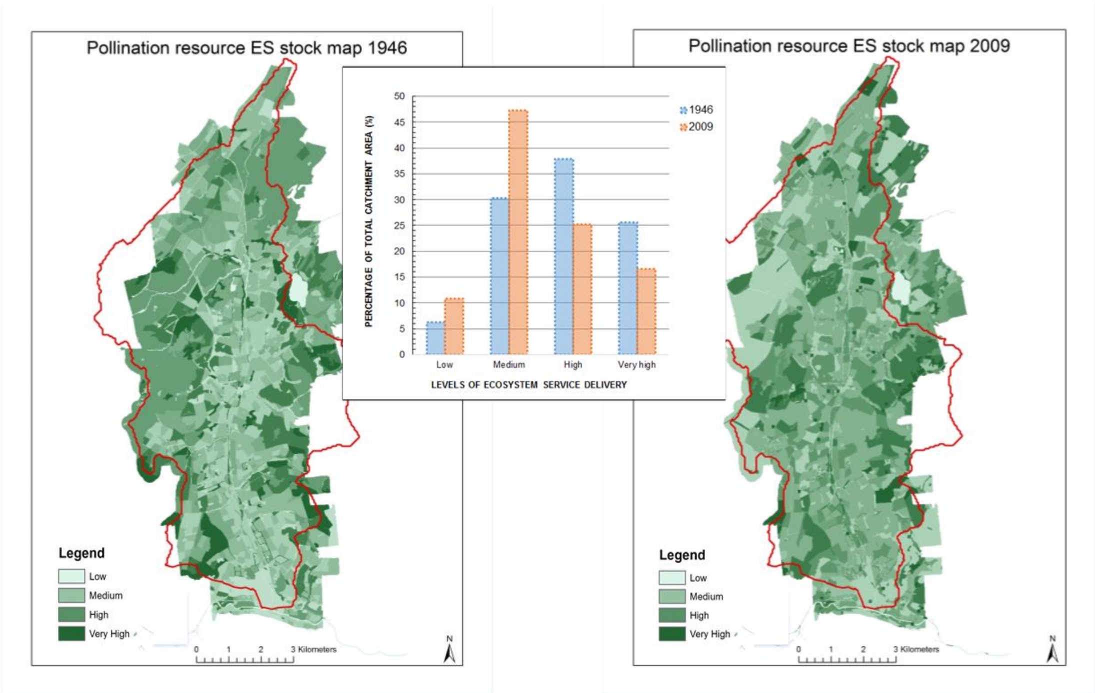

Figure 4-27: Pollination resource supply capacity in the Eddleston catchment

In the Eddleston, medium supply areas accounted for about 30% of the total catchment area in 1946 while the high supply areas accounted for about 38% of the total catchment area during the same period and in total these accounted for about 81% of the total catchment area (Figure 4-27). By 2009, medium supply areas accounted for about 48% of the total catchment area.

Water quality regulation (regulating ES)

Water quality regulation refers to the role of habitats and their linkages to vegetation structure, soil type and other catchment processes to filter and take up pollutants; contributing to water purification.

The maps for both catchments show that the capacity to regulate water quality was higher in 1946 than in 2009.

In the Ale catchment, high water quality regulation areas in 1946 were predominately in the upper catchment (uplands), with some high supply areas in the lower catchment (figure 4-28).

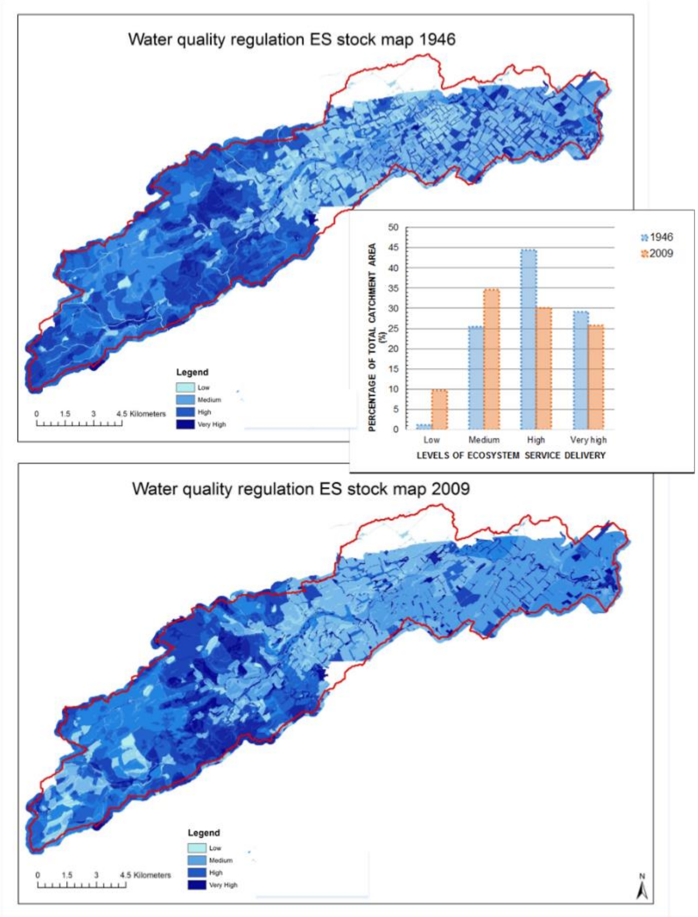

Figure 4-28: Water quality regulation areas in the Ale catchment

Similarly, the uplands of the Eddleston and the top of this catchment had a high water quality regulation capacity in 1946.

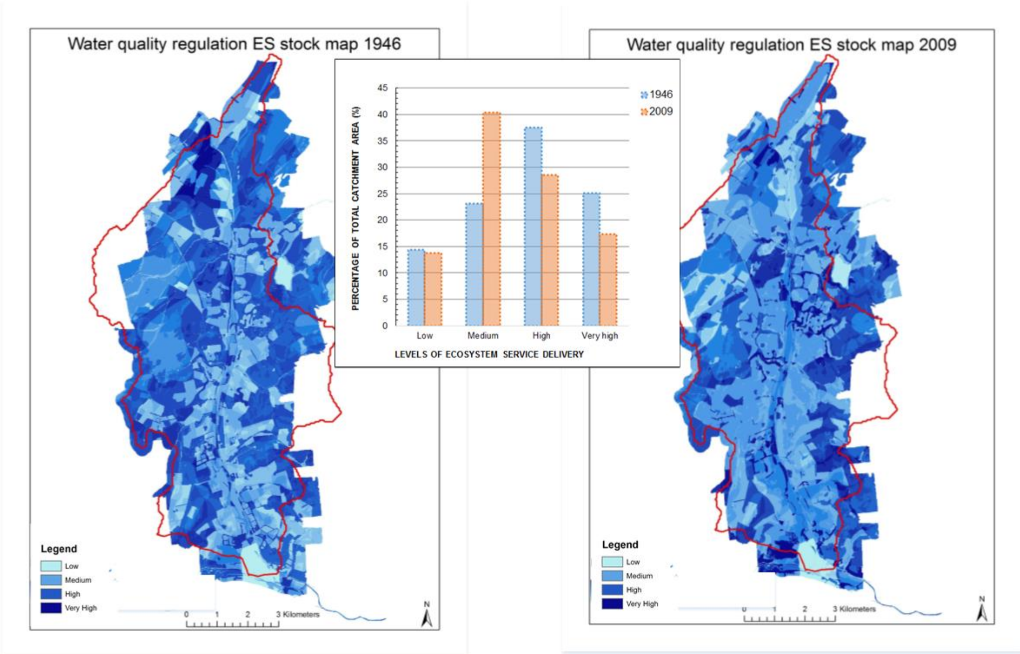

Figure 4-29: Water quality regulation areas in the Eddleston catchment

High water quality

regulation areas were associated with semi-natural habitats dominant in these catchments at that time i.e. wet bogs and blanket bogs, heathland, broadleaved woodland and hedgerows. Following the reduction of these habitat types, in 2009 the water quality regulation capacity also reduced from high to medium capacity levels. However, there were few hotspots associated with the remaining semi-natural habitats with a high water quality regulation capacity, especially in areas with woodland plantations. On the other hand, low water quality regulation areas in 2009 were associated with arable areas, improved grassland areas and recently felled coniferous woodland, as the use of agro chemicals in these areas contributes to water pollution. The drainage of bogs also reduced their capacity to retain and filter water and hence the remaining modified bogs found in these catchments have reduced capacities to purify water.

The bar graphs show that in 1946 nearly 40% of the catchments` total areas had a high capacity to regulate water quality. By 2009, there had been a shift to medium supply capacities in both catchments, with about 35% and 40% of the Ale and Eddleston catchment areas respectively having a medium capacity to provide this ES.

Crop production (provisioning ES)

This refers to the production of crops such as cereals which are cultivated and harvested for human consumption or for other uses like producing stock feed or use in the distilling or malting industry. As mentioned earlier, the crop production capacity in the study catchments was approximated using changes in area under arable land between 1946 and 2009, coupled with the crop production yield estimates for Scotland for the respective two dates. According to the Scottish Government annual agricultural census reports, cereals notably, barley and wheat are

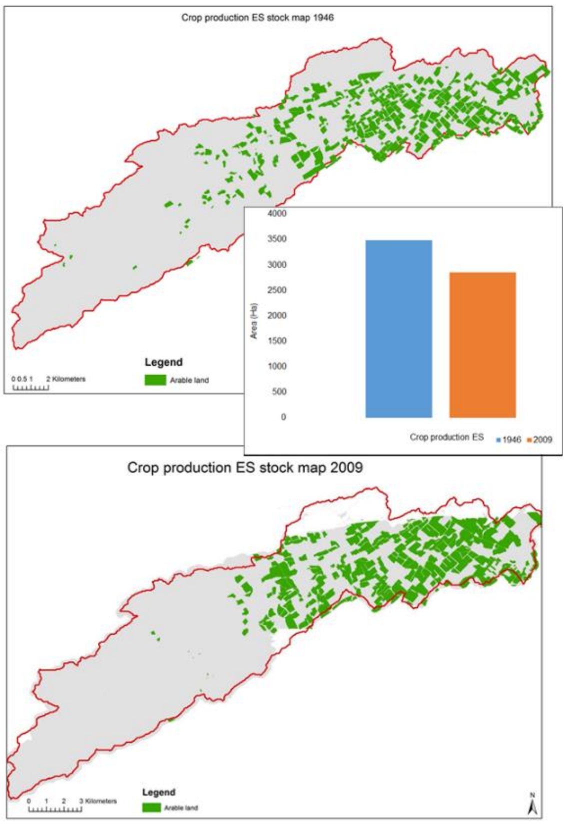

Figure 4-30: Crop production areas in the Ale catchment

the main crops grown in the Scottish borders.

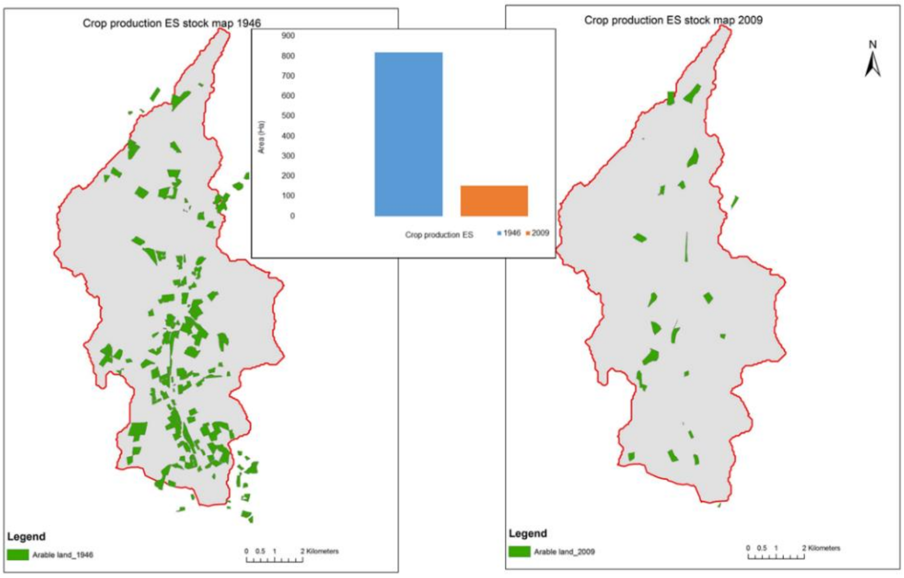

Figure 4-31: Crop production areas in the Eddleston catchment

The maps and bar graphs (figures 4-30 and 4-31) show that area under arable land had slightly decreased in the Ale catchment by 2009 while it had significantly declined in the Eddleston catchment. These reductions in area under arable land were interpreted as indications of a decline in crop production in these catchments. Crop production estimates in Scotland show that annual crop production of oats, wheat and barley declined from over 800 000 tonnes in the 1940s to less than 120 000 tonnes since the 1990s (UK NEA, 2011). However, during this period there was an increase in mean annual yields owing to technological and scientific advances, including the use of fast growing varieties.

Land/soil erosion risk (regulating ES)

Erosion risk refers to the susceptibility of land to soil erosion. This is a function of vegetation cover, topography, human activities and underlying geology.

The maps presented here show changes in catchment areas that were vulnerable to soil erosion in 1946 and 2009.

Figure 4-32 shows that the land/soil erosion risk had increased in the Ale catchment by 2009 while it decreased in the Eddleston catchment (Figure 4-33).

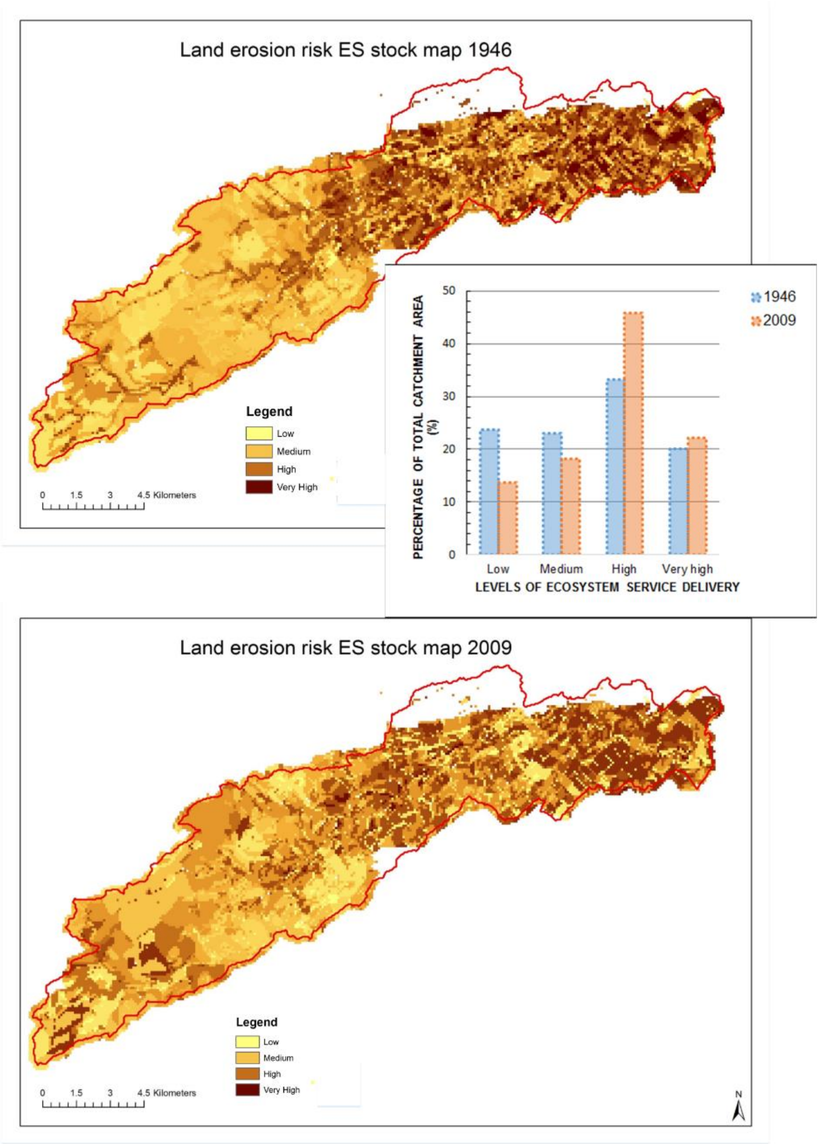

Figure 4-32: Soil erosion risk in the Ale catchment

In 1946, soil erosion was mainly confined to the lower parts of the Ale catchment, dominated by arable land and farming activities which destabilise soil, while about 23% had a low soil erosion risk. By 2009, the soil erosion risk had spread to the upper catchment and was mainly associated with recently felled coniferous woodland plantation areas. As such, in 2009 the greater proportion (about 50%) of the total catchment area was classified as having a high land/soil erosion risk.

In contrast, in the Eddleston catchment, the soil erosion risk reduced from a high capacity in 1946 to a medium capacity in 2009. In 1946 about 40% of the total Eddleston catchment area had a high soil erosion risk. As shown in the 1946 map, the soil erosion risk was dominant throughout the catchment, including the upland marginal areas on either sides of the Eddleston valley and within the Eddleston valley.

By 2009, the soil erosion risk had reduced and the greater proportion (45%) of the catchment area had a medium soil erosion risk. Such a reduction in the soil erosion risk could be associated with the massive reduction in arable land within the Eddleston valley by 2009. However, regions of high soil erosion risk were still evident in 2009, some of which included areas of recently felled coniferous woodland plantation areas.

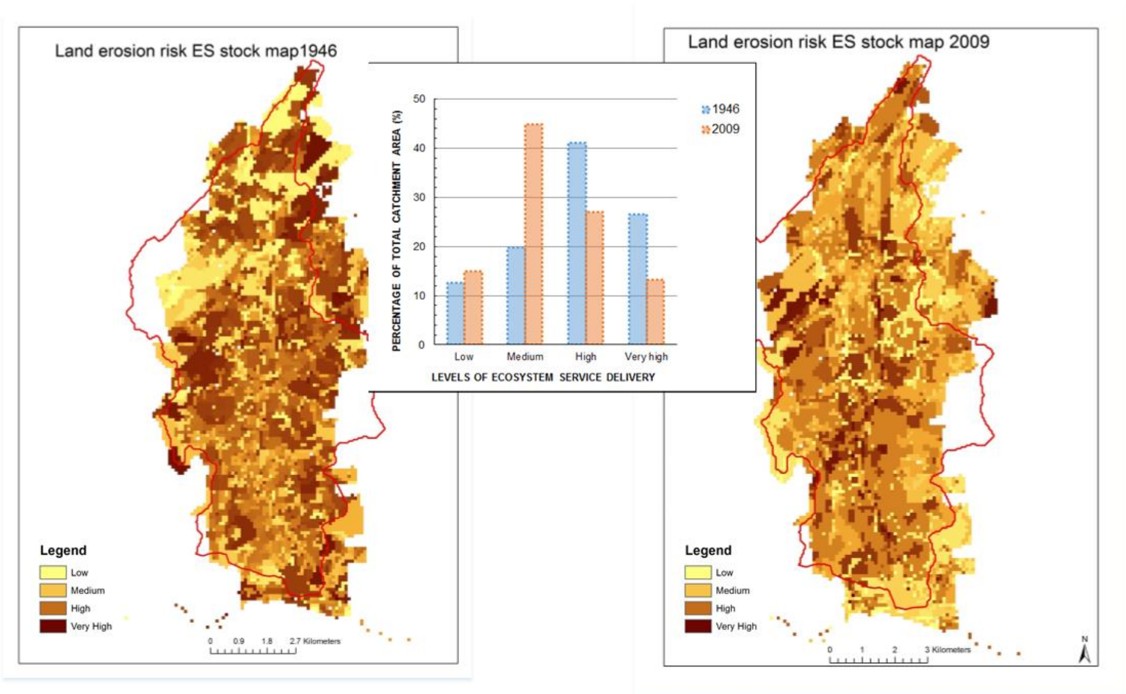

Figure 4-33: Soil erosion risk in the Eddleston catchment

The difference in the soil erosion risk in these catchments could be due to the extent of arable land reduction in the Eddleston by 2009. In comparison, arable land remained dominant in the Ale catchment and hence the soil erosion risk also remained high. The increase in human activities such as conifer harvesting in the uplands will also have contributed to the soil erosion risk in 2009, as this destabilises the soil exacerbated by variations in relief in the uplands.

4.3.3 Section summary

Assessment of changes in ecosystem service delivery in the study catchments between 1946 and 2009 were based on: (a) the spatial location and variations in ecosystem service supply areas, as shown on the ecosystem service maps, (b) a qualitative assessment of changes in ecosystem service supply levels (low, medium, high) in accordance with the SENCE methodology used to map ecosystem services in this study, and (c) the extent of ecosystem service supply areas between 1946 and 2009 derived from the zonal frequency counts on the ecosystem service maps.

Results show that the upland areas of the study catchments had a reduced capacity to supply biodiversity, pollination resource, water quality regulation and soil carbon storage ecosystem services by 2009, associated with reduction in semi-natural habitat types during the period since 1946. In contrast, the low lying areas of the study catchments had an increased potential to supply provisioning services especially livestock production.

The presented ecosystem service maps also show spatial overlaps among ecosystem services. For example, the increase in woodland plantations by 2009 also led to increased capacities for timber provision, vegetation carbon storage and flood regulation ecosystem services. Similarly, the reduction of semi-natural habitat types like bogs also influenced the reduction in high supply areas of biodiversity, soil carbon storage, water quality regulation and pollination resource ecosystem services. Also shown in the ecosystem service maps was the presence of high ecosystem service hotspots associated with remaining semi-natural habitat patches by 2009.

Other observed changes were in spatial trade-offs from one ecosystem service type to another. Such a trade-off was between livestock production and crop production as the area under arable land in 1946 was taken over by improved grassland.

The conceptual figures (Figure 4-34, 4-35 and 4-36) below illustrate identified shifts in levels of ecosystem service delivery between the two dates.

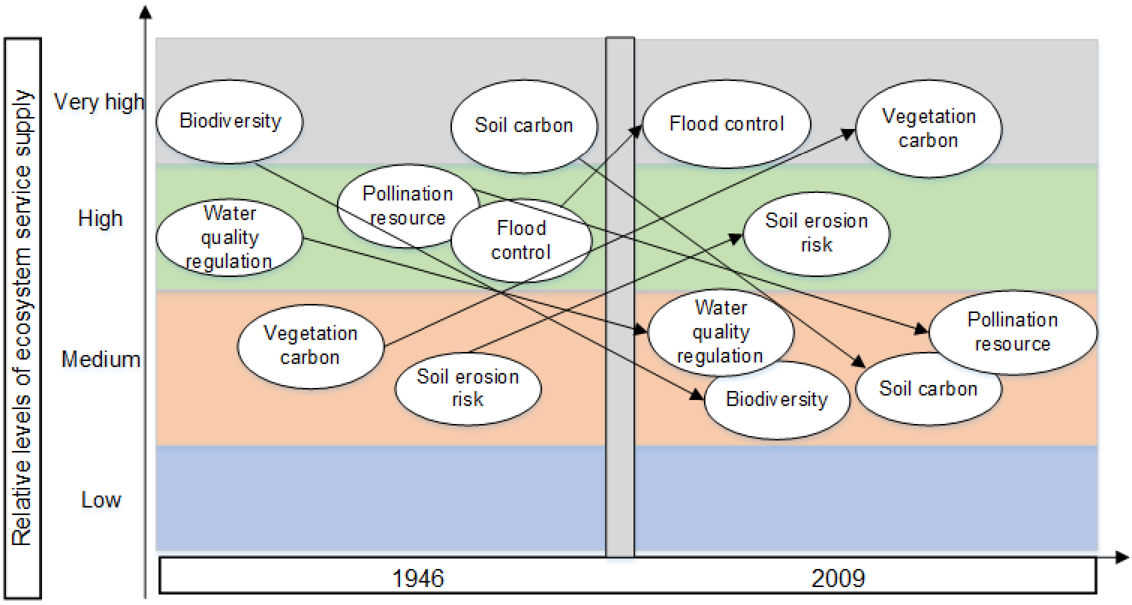

Figure 4-34: Changes in the levels of ecosystem service supply in the Ale catchment

Figure 4-34 shows that most regulating ecosystem services (biodiversity, soil carbon storage, water quality regulation and pollination resource) shifted from higher supply capacity levels in 1946 to lower capacities by 2009. In contrast, flood control and vegetation carbon supply levels shifted to higher supply capacities during the same period. Alongside this was also an increase in the soil erosion risk from lower to higher levels by 2009.

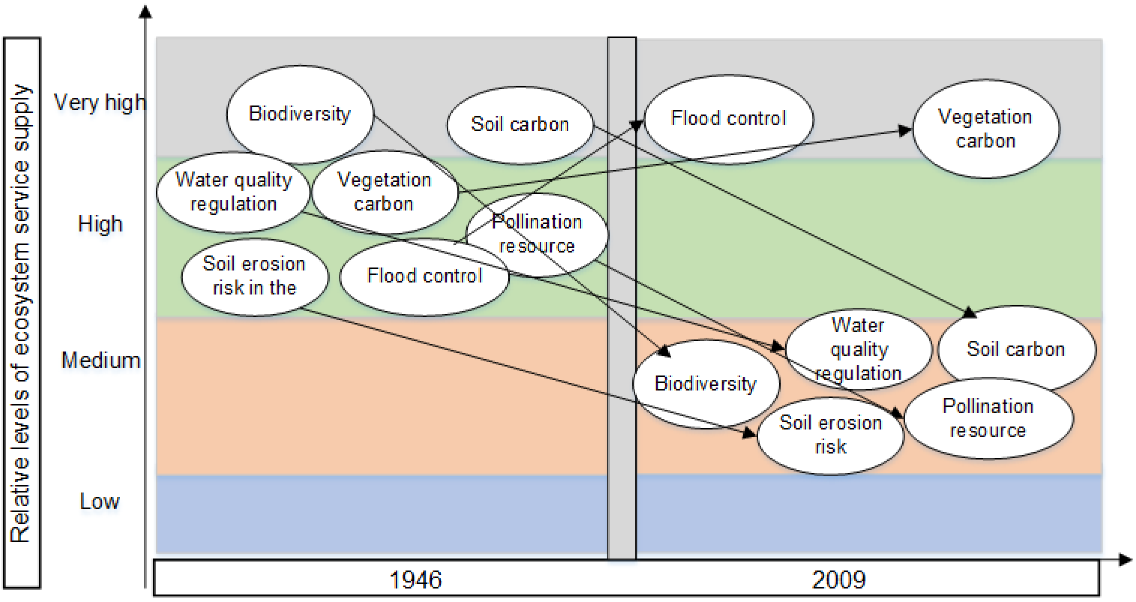

Figure 4-35: Changes in the levels of ecosystem service supply in the Eddleston catchment

Similar to the Ale catchment, ecosystem service delivery levels in the Eddleston catchment shifted from higher supply capacities in 1946 to lower capacities in 2009, save for flood control and vegetation carbon ecosystem services (Figure 4-35). However, the soil erosion risk shifted from higher levels to lower levels in this catchment compared to the Ale catchment.

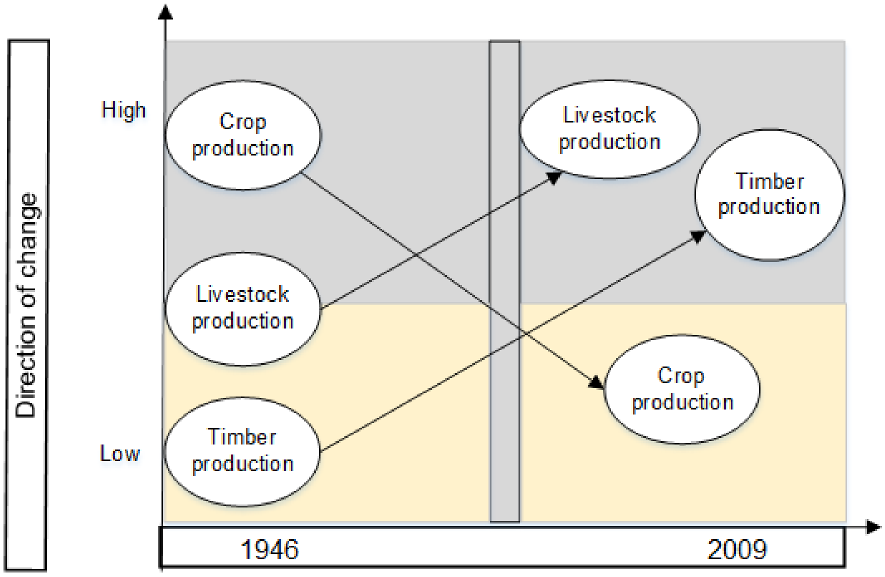

Figure 4-36: Changes in supply capacities for provisioning ecosystem services in the Ale and Eddleston catchments

As mentioned earlier, changes in area under coniferous woodland, arable land and improved grassland between 1946 and 2009 was used as an indicator of their capacity to supply timber, crop and livestock ecosystem services respectively. Figure 4-36 shows changes in the supply capacities of these ecosystem services in both catchments. Both catchments recorded an increase in the capacity for timber and livestock production while crop production decreased by 2009.

4.4 Results chapter synthesis

4.4.1 Influence of habitat changes on ecosystem service delivery

The aim of this chapter was to demonstrate spatio-temporal changes that have occurred to habitats and ecosystem services in the study catchments between 1946 and 2009. Such spatial changes were based on analysing changes in the extent, patterns and location of habitats and ecosystem services between the two dates. This has provided an insight into the state of habitats prior to the onset of key drivers, such as the onset of agricultural intensification, which greatly influenced habitat and ecosystem service changes in the catchment landscapes in the 1940s. It has also demonstrated how habitat changes influenced ecosystem service delivery over time. However, the changes that occurred to these between 1946 and 2009 are not captured as this analysis was based on two specific dates and did not track variations between these dates.

The findings show that both catchments depicted broadly similar habitat and ecosystem service changes, with variations influenced by the difference in catchment sizes, catchment topography and extent of habitat changes. This could be suggestive of similar drivers of change in these catchments. Such drivers of change led to the increased dominance of intensively managed habitat types by 2009 and a reduction in semi-natural habitats. This in turn influenced the delivery of ecosystem services associated with these habitat types. Consequently, by 2009 the study catchments had increased capacity to supply provisioning services i.e. livestock production and timber, and regulating ecosystem services of vegetation carbon and flood regulation associated with increased woodland and improved grassland. In contrast, there was a reduction in the supply of biodiversity, soil carbon storage, pollination resource and water quality regulation associated with decrease in bogs and other semi-natural habitat types. The figure below illustrates the influence of identified habitat changes on ecosystem service delivery.

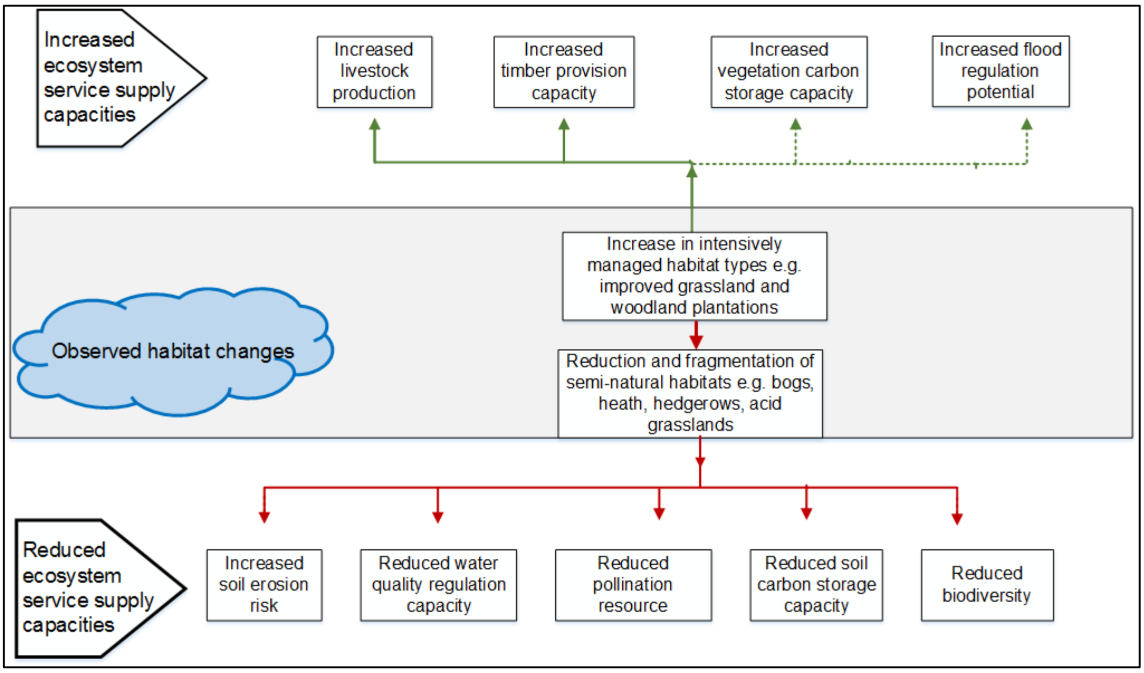

Figure 4-37: Influence of habitat changes on ecosystem service delivery

As illustrated in figure 4-37, the increase in intensively managed habitat types led to increased supply capacities for provisioning ecosystem services while constraining the supply of most regulating and supporting ecosystem services. The increase in woodland plantations areas also led to increased capacity for vegetation carbon storage and flood regulation ecosystem services.

This means a change in one habitat type leads to multiple ecosystem service changes. So, while converting one habitat type such as upland bogs to coniferous woodland, was a “single” habitat change, this led to multiple ecosystem service changes in the study catchments (in soil carbon storage, biodiversity, pollination etc.). These findings reflect the multifunctional role of semi-natural habitats in ecosystem service delivery, as reported in other studies. For example, Burkhard et al. (2012) observed that natural to semi natural land cover types (e.g. peatlands, moors and heathlands) had a high capacity to supply several ES, while intensively managed habitat types (e.g. improved grassland) have very low capacities to provide multiple ecosystem services. Similarly, Vrebos et al. (2015b) observed that natural vegetation provided more ecosystem services than intensively managed land uses, while Crouzat et al. (2015), concluded that heterogeneous landscapes provide richer sets of ecosystem services than homogenous ones.

On this basis, it can be argued that the multifunctional role of both study catchment landscapes changed over time into intensively managed ones, with a higher capacity to provide provisioning ecosystem services, both in the low lying and upland areas. Mastrangelo et al. (2014), define landscape multifunctionality as "the capacity of a landscape to simultaneously support multiple benefits to society from its interacting ecosystems". These authors argue that multifunctional landscapes are associated with a high potential to supply regulating ecosystem services. As indicated by the Shannon Diversity Index, results from the Ale and Eddleston catchments show that the 1946 catchment landscapes were more diverse and heterogeneous compared to 2009, depicting less simplified and multifunctional landscapes. Simplification of catchment landscapes by 2009, can be argued to have impacted on their capacity to supply multiple ecosystem services compared to 1946. This reflects the increasing pressure of human activities on catchment landscapes over time.

4.4.2 Impact of habitat fragmentation on ecosystem service delivery

The net change habitat maps (Figure 4-11 and 4-13), showed that the uplands of the study catchment were mostly fragmented, with reduced semi-natural habitats connectivity and mean patch sizes. Changes in the configuration of these habitat patches across space also influenced changes in ecosystem service supply areas. Figure 4-38, is an extract from the uplands of the Ale catchment, illustrating the impact of spatial habitat changes on ecosystem services.

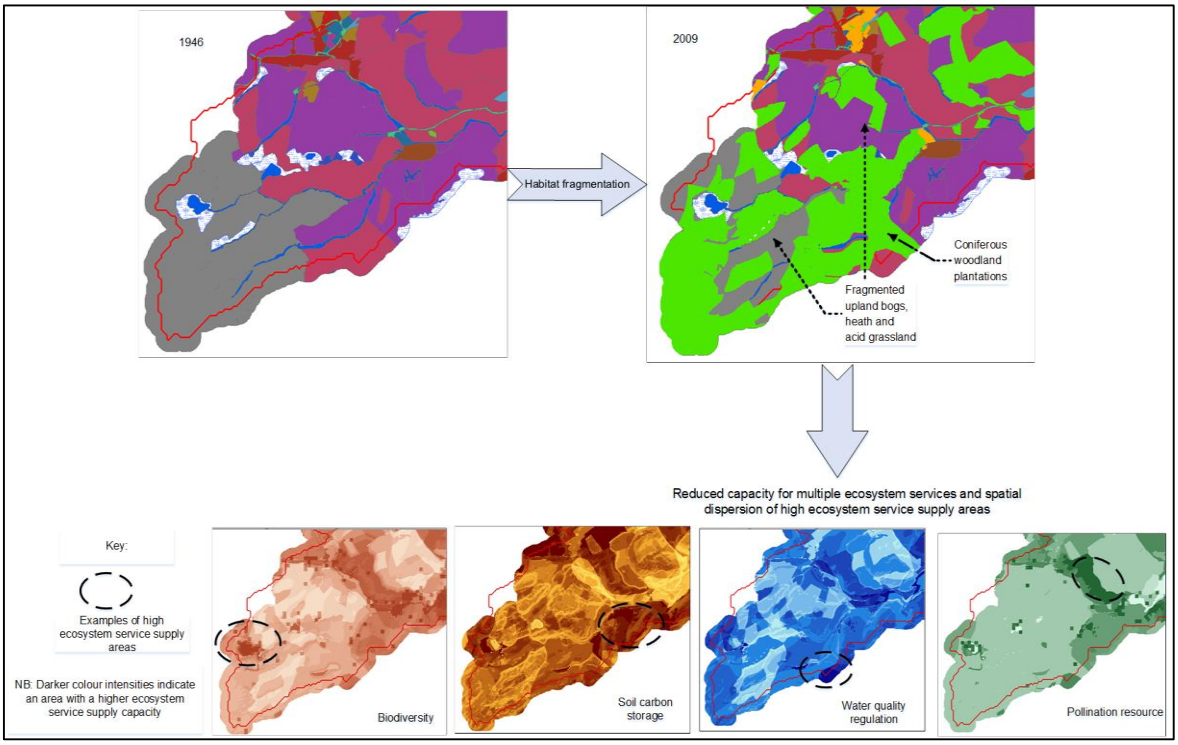

Figure 4-38: Impact of habitat fragmentation on ecosystem service delivery

Figure 4-38 shows that the introduction of coniferous woodland plantations, a new habitat type in the uplands of the study catchments, led to fragmentation and reduced connectivity between semi-natural habitat patches. While such a change reduced the capacities for high multiple ecosystem service delivery, it also resulted in spatial dispersion of high ecosystem service supply areas and “hotspots” for these were associated with remaining semi-natural habitat patches, as indicated by circled dark colour intensities on the ecosystem service maps. On the other hand, areas occupied by coniferous woodland plantations show a reduced capacity for the supply of these ecosystem services, referred to, as ecosystem service “cold” spots by others such as Queiroz et al. (2015).

This means changes in spatial configuration of semi-natural habitats in the study catchments by 2009 led to increase of regulating and supporting ecosystem service “cold” spots, in favour of timber provisioning ecosystem service. Such changes have catchment wide impact on ecosystem service delivery. As noted by Bunce et al. (2014), planting of coniferous woodland in the uplands increases the runoff risk and alters the catchment hydrology, as well as modifying the drainage and hydrology of bogs. It also impacts on species population dynamics (Fahrig, 2007) and hence biodiversity ecosystem service which, as a supporting ecosystem service impacts on the delivery of multiple ecosystem services (Mace et al., 2012).

Habitat fragmentation reduces habitat connectivity, which impacts on species population dynamics, including their population sizes, movement, distribution and migration (Soons et al., 2005, Fahrig, 2007). These authors note that those species whose occurrence is primarily dependent on habitat patch size are mostly affected by the reduction in habitat patch sizes. As habitats of the same type become isolated, species exchange and movement between these becomes difficult as some species are not able to travel long distances between widely separated remaining habitat patches (Andrén, 1994). Such movement could be for regular needs like searching for food, shelter or to access a seasonal habitat for migratory species e.g. birds (Fahrig, 2007). In the process, such species become isolated and their movement restricted, especially if faced by introduced barriers such as expansive coniferous woodland plantations; unsuitable for their needs. Habitat fragmentation could subsequently lead to mortality of vulnerable species, species population declines or local extinctions (Antwi et al., 2008, Fahrig, 2003, Fahrig, 2007) with an ultimate risk of increased biodiversity loss.

In contrast, in the low lying agriculture dominated catchment areas, mean habitat patch sizes and proximity index for intensively managed habitat types had increased by 2009. This is attributed to a reduction of hedgerows to increase field sizes and ploughing on semi-natural grasslands. These changes, as already discussed also impacted on multiple ecosystem service delivery capacity of these areas.

4.4.3 Ecosystem service trade-offs and synergies across space and time

The increase in intensively managed habitat types such as improved grassland and coniferous woodland in 2009, can be interpreted as a preference and prioritisation of provisioning ecosystem service while regulating and supporting services were impacted upon. Rodríguez et al. (2006) defines a trade-off as, "*the intentional or unintentional elevation in the provision of one ecosystem service at the expense of another ecosystem service*". These authors note that intentional trade-offs are related to the drive to meet human needs like food provision while unintentional trade-offs could result from lack of knowledge about other ecosystem services and their interactions, especially the less tangible ones which do not have explicit markets.

Mackey et al. (1994) note that coniferous woodland plantations were introduced in most upland areas as these were considered marginal areas that were poor for agricultural productivity. In the process semi-natural upland habitat types especially bogs, heathland and acid grassland mosaics etc. were either interspersed or replaced by coniferous woodland plantations (Haines-Young et al., 2000). Following Mackey et al. (1994)`s observation, this reflects the probably unintentional trade-offs related to lack of knowledge and or lack of interest in their value at that time. Provisioning ecosystem services were prioritised at that time and areas occupied by semi-natural habitats such as bogs were viewed as wastelands. These, however, were important in the supply of other intangible, less obvious but arguably of equal importance for human survival. Trade-offs between provisioning and regulating ecosystem services are common and have been acknowledged by a number of commentators e.g. Raudsepp-Hearne et al. (2010), Haines-Young et al. (2012), Maes et al. (2012b), Jopke et al. (2015), Science for Environment Policy (2015) and Queiroz et al. (2015).

As shown in the habitat transition matrices (Table 4-2 and 4-3), most semi-natural habitat types were converted into intensively managed habitat types, notably coniferous woodland plantations and improved grassland. This further illustrates spatial trade offs in ecosystem services. Most of the areas that remained unchanged between these dates were occupied by improved grassland in 1946 and remained so in 2009. Over 70% of area under improved grassland in 1946 remained so in 2009 in both catchments. A few habitat types such as excavation and dug up sites that were present in the Ale catchment in 1946 were no longer there in 2009 as over 60% of these areas were replaced by improved grassland. On the other hand, the introduction of other intensively managed habitat types such as the open cast mine in the Eddleston catchment in 2009 replaced semi-natural habitat types such as acid grassland, bogs and cultivated arable land.

These findings show that trade offs in ecosystem services have led to declines in regulating ecosystem services over time due to both human induced and ecological processes. Such patterns of tradeoffs in ecosystem services have also been noted elsewhere. For example, in the Lower Yangtze River Basin in China, agriculture intensification and industrial development intended for alleviating rural poverty led to marked losses of regulating ecosystem services (Dearing et al., 2012).

The increase in coniferous woodland plantations in the uplands increased supply capacities for vegetation carbon and flood regulation ecosystem services in addition to timber provision. This may be suggestive of ecosystem service bundles and/or synergies in ecosystem service provision, as the 2009 maps for these ecosystem services showed spatial overlaps with coniferous woodland plantation areas in the uplands. Raudsepp-Hearne et al. (2010) define ecosystem service bundles as a set of ecosystem services that repeatedly appear together across space and time, while a synergy is the elevation of one ecosystem service caused by an increase in another ecosystem service. Synergies for timber stock and other ecosystem services like air quality regulation, and vegetation carbon storage have also been reported in other studies e.g. Jiang et al. (2013) and Bunce et al. (2014).

The increase in vegetation carbon storage supply capacity seen by 2009 can also be perceived as a positive impact from the increase in coniferous woodland. It may also have somewhat compensated for the reduced soil carbon storage capacity by 2009, following the loss of semi-natural habitat types such as bogs which had a very high capacity for soil carbon storage in 1946. In spite of these possible positive impacts associated with increased coniferous woodland plantations, it may still be argued that this brought more harm than good. For example, harvesting and removal of mature coniferous woodland trees would interfere with continued provision of vegetation carbon and flood control. Increased human activities during planting, harvesting, the use of machinery and cutting of these trees would instead contribute to increased soil erosion risk (Bunce et al., 2014). This was reflected in the soil erosion risk maps which showed increased soil erosion risk in the uplands of the study catchments in 2009 compared to 1946.

4.4.4 Differences in extent of ecosystem service delivery between the two study catchments

This study mapped and assessed ecosystem services changes in two selected study catchments. Although both study catchments depicted broadly similar patterns of habitat and ecosystem service changes the extent of such changes differed between the two catchments. The Eddleston catchment for example, had a lower Shannon diversity Index compared to the Ale catchment in 2009. The Eddleston catchment also had less than 2% of its total catchment area under arable land in 2009, it recorded an increase in built land due to expansion of Peebles and Eddleston Villages and the introduction of an open cast mining site. In the Ale catchment by contrast, arable land was in 2009 among the most widespread habitat types, accounting for about 15% of the total catchment area. The Ale catchment also did not record the introduction of new built up areas, save for expansion of roads and farm houses.

While the study catchments also differ in sizes, the differences in the extent of habitat changes can be attributable to varying land use preferences and choices in these catchments over time, as well as land ownership. For example, the marked reduction in area under arable land in the Eddleston catchment in 2009 indicates spatial trade-offs between crop and livestock production. This could be suggestive of local farmer preferences for livestock production in this catchment. Such an understanding is important in local stakeholder involvement and management intentions should consider such different priorities in catchment management.

5 Discussion

5.1 Chapter introduction

The previous chapter presented findings from analysis of spatio-temporal changes in extent, distribution and location of both habitats and ecosystem services in the study catchments. In this chapter, limitations of current proxy based approaches to mapping ecosystem services, which should be taken into account in interpreting findings and conclusions from this study are first discussed. Secondly, the major factors that have influenced habitat and ecosystem service changes in the study catchments are discussed. The last section of this chapter is a discussion on the implications of this study in catchment management.

5.2 Current approaches in mapping ecosystem services

This section discusses and reflects on the limitations of current proxy based approaches to mapping ecosystem services which should be taken into account in interpreting conclusions from this study. As discussed in the literature review chapter, mapping and assessment of changes in ecosystem services is one of the important elements of the ecosystem service concept considered crucial in moving this concept from theory into practice (Burkhard et al., 2012). As an emerging research area, the lack of independent ecosystem services data means that current practices mainly rely on proxy based approaches to mapping ecosystem services. Of these, the use of habitat/LC data is currently the most common proxy based approach to mapping ecosystem services (Andrew et al., 2015, Seppelt et al., 2011). Authors such as Burkhard et al. (2009) acknowledge that habitat/LC based proxy data provide an appropriate base for mapping ES. While this is understandably so, there are limitations (Eigenbrod et al., 2010a, Schulp et al., 2014) over the use of these proxy based approaches to mapping ecosystem services.

Limitations in current practices relate to simplifying approaches in mapping ecosystem services which include: (1) the assumed linear relationship between habitat changes (quality and quantity) and ecosystem service delivery, and (2) use of relative scores (informed by expert opinion and scientific literature) and look up tables to indicate relative levels of importance of different habitat/land cover types in ecosystem service delivery. While this is widely applied in current land cover based ecosystem service mapping approaches, there are questions and criticisms levelled against such simplifying approaches, which this study might also be prone and amenable to. For example, there are questions related to linearity or otherwise in any loss of ecosystem services as habitats deteriorate (in quantity, quality or location) – or indeed gains. Yet ecosystems are complex and their processes, functions and interactions involve nonlinear dynamics (Koch et al., 2009). There are also issues of uncertainty e.g. tipping points, some of which are irreversible when certain thresholds are reached (Leh et al., 2013) which such simplifications do not adequately capture.

Expert informed relative scores of low, medium, high etc. are noted to introduce inconsistencies and subjectivity as different experts can have different weights on levels of ecosystem service supply (Andrew et al., 2015, Verhagen et al., 2015). Look up tables, on the other hand are a source of generalisation error (Eigenbrod et al., 2010a) as they assume that the level of ecosystem service delivery is constant across habitat/LC classes yet there are variations in species composition, structure and age of habitat types etc. which all influence their ability to supply ecosystem services.

By contrast, while the aforementioned are acknowledged as a general limitation to current ecosystem service mapping practice, such simplification is justified by the fact that current scientific knowledge and understanding on ecosystem processes, functions and ecological interactions remains incomplete (Vermaat et al., 2015). Such understanding has not reached levels were exact figures/quantifications can be assigned to ecosystem services (Nedkov and Burkhard, 2012). Rather the knowledge that exists is sufficient to provide indicative scales on ecosystem service delivery and necessitates the use of simplifying approaches and models to spatially represent and assess ecosystem services.

This study qualitatively assessed the importance of different habitat types in ecosystem delivery using relative scales, informed by expert opinion and literature review. Such an assessment of using relative scales of low, medium, high etc. assesses the capacity of a habitat and hence its importance in supplying an ecosystem service. In so doing, key habitat types associated with delivering ecosystem services can be identified and provide a basis for comparison across space and time the role played by different habitats and the ecosystem services they can potentially supply.

Improvements are also being made towards providing further detail in mapping ecosystem services as there is increasing recognition of the influence of other spatial factors like soil type, topography in ecosystem delivery (Nemec and Raudsepp-Hearne, 2013). The SENCE method used in this study, for example, integrated habitat maps with other spatial data such as soils maps, topography etc. to capture the influence of spatial variations in ecosystem service delivery.

Simplifying approaches are perceived by others e.g. Verhagen et al. (2015) as important in communicating with non-specialists e.g. stakeholders and policy makers, the role of ecosystems and complexities associated with ecosystem functions and processes in ways they can understand, which they argue is the crux of the ecosystem service concept. By mapping ecosystem services ecological complexities are simplified (Hauck et al., 2013a) and stakeholders and policymakers can be engaged in discussions in ways that can visualise, easily relate to and conceptualise the link between ecosystems and how they contribute to human well-being. Plant and Ryan (2013) consider this as a unifying language that simplifies the, at times difficult to communicate habitat/LC terminologies.

However, accuracy of ecosystem service maps and their validation remains an insufficiently addressed shortfall in current ecosystem service mapping practices (Schulp et al., 2014, Willemen et al., 2015). Their accuracy currently depends on the accuracy of proxy data and methods used to map these. Others such as Eigenbrod et al. (2010a) have criticised the use of such proxies as they are noted to contribute to errors in ecosystem service maps. Andrew et al. (2015) also note that this might present a challenge in operationalising the ecosystem services concept.

Despite these current limitations proponents of habitat/land cover based approaches to mapping ecosystem services like Burkhard et al. (2009) assert that the use of proxies especially where there are data limitations is better than ignoring such ecosystem services altogether. Nemec and Raudsepp-Hearne (2013) observed that, in current practice, importance in mapping ecosystem lies on the end goal of the mapping purpose rather than the consistence in indicators, tools and approaches used. Willemen et al. (2015) is of the view that best ecosystem service mapping practices need to be robust, transparent and stakeholder relevant to support decision making.

Verhagen et al. (2015) suggests that credibility of ES maps can be assessed by comparing maps of similar ecosystem services derived by different methods or tools. A study done by Vorstius and Spray (2015) in the Eddleston catchment, one of the study catchments in this research, compared ecosystem service maps produced using InVEST, SENCE and EcoServ-GIS. These authors observed that, the SENCE method included more detail, data sets and captured local variations compared to the other two tools. Based on this, the ecosystem service maps generated in this study can be argued to be credible. Perhaps of fundamental importance is also the stakeholder relevance of the generated ecosystem service maps as the 2009 ecosystem service maps were used by the Scottish Borders Council for stakeholder consultations in the study catchments during the LUS pilot project (Spray, 2014) and in the process were verified and validated by stakeholders.

5.3 Major drivers of habitat and ecosystem service changes

The final research question posed in this study was to identify major factors which have influenced habitat and ecosystem service changes in the study catchments. Both study catchments depicted broadly similar patterns of habitat and ecosystem service changes, suggestive of similar drivers of change. As illustrated in the conceptual framework underlying this study (Figure 5-1), drivers of change are a reflection of human actions that influence habitat “ecosystem” changes directly or indirectly. In so doing these impact on ecosystem processes, functions and ultimately on ecosystem service delivery.

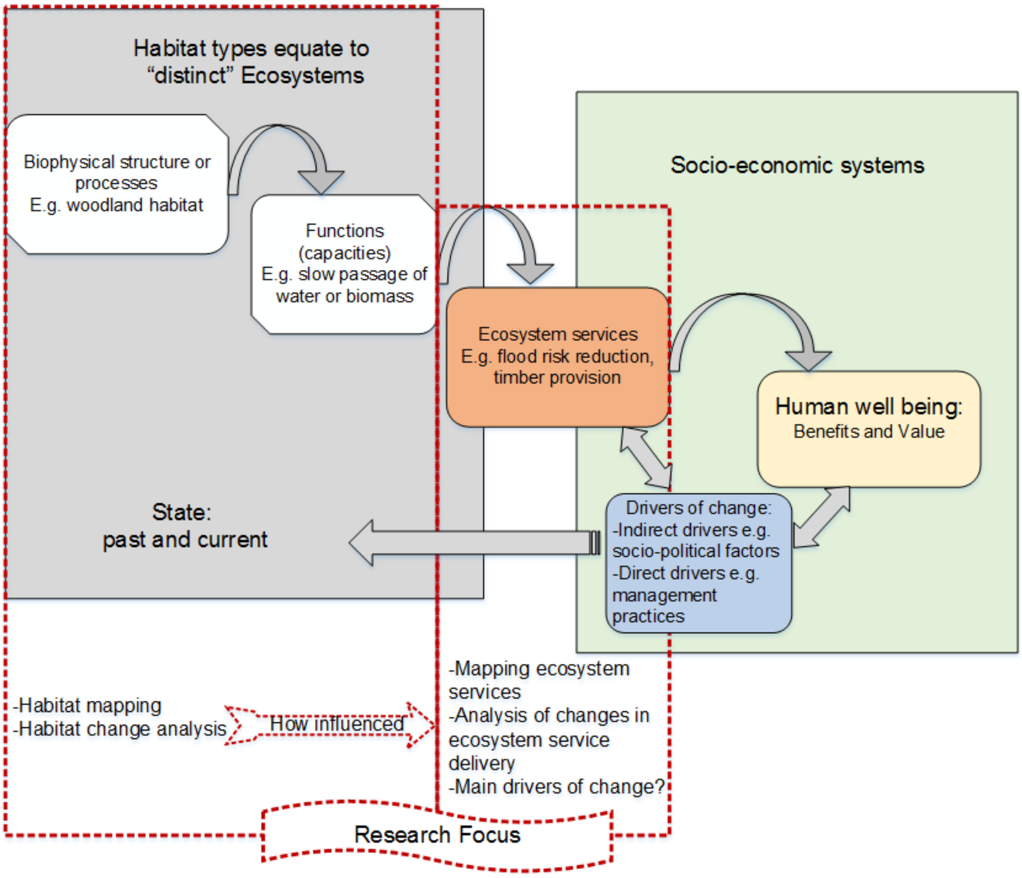

Figure 5-1: Ecosystem services cascade conceptual framework
(*Source: Modified from Haines-Young and Potschin (2010)*)

Since the Second World War, socio-political factors including changes in policies have been the main drivers of change influencing landscape and habitat changes in the Scottish countryside (Firbank et al., 2013, Miller et al., 2009). Alongside these were other drivers of change related to economic factors, technological advancements, demographic changes, and rural development, some of which were triggered by these policies. These drivers of change are closely linked, have acted interdependently and at times simultaneously at varying levels, as will be discussed later. The habitat and ecosystem service changes identified in the study catchments were linked to these major factors that have influenced habitat and landscape changes elsewhere in the Scottish countryside. However, the impact and extent of these drivers of change differed between and within the study catchments. The sections below discuss these phases of policy shifts, including how these were linked to or influenced other drivers of change.

Influencing policies range from international conventions and European level directives to UK and Scottish level legislation and policies. Figure 5-2 below illustrates how identified direct and indirect drivers of change influenced habitat and ecosystem service changes. Two phases of policy shift have been identified: the first phase covers the period between the 1940s and 1980s, when legislation and policies were introduced to promote agriculture intensification and afforestation policies (Bunce et al., 2014, Bowers, 1985). The second phase was the period after the 1980s (to 1990s) marked by a shift and adjustment in these drivers of change to include legislation and policies aimed at land use diversification, biodiversity conservation and environmental management, including the promotion of sustainable agriculture practices (Sutherland, 2002, Robertson and Swinton, 2005). This phase has continued into the start of the 21st century, with most recently the ecosystem services concept gaining increased interest as a possible framework for the management of the natural environment (Guerry et al., 2015).

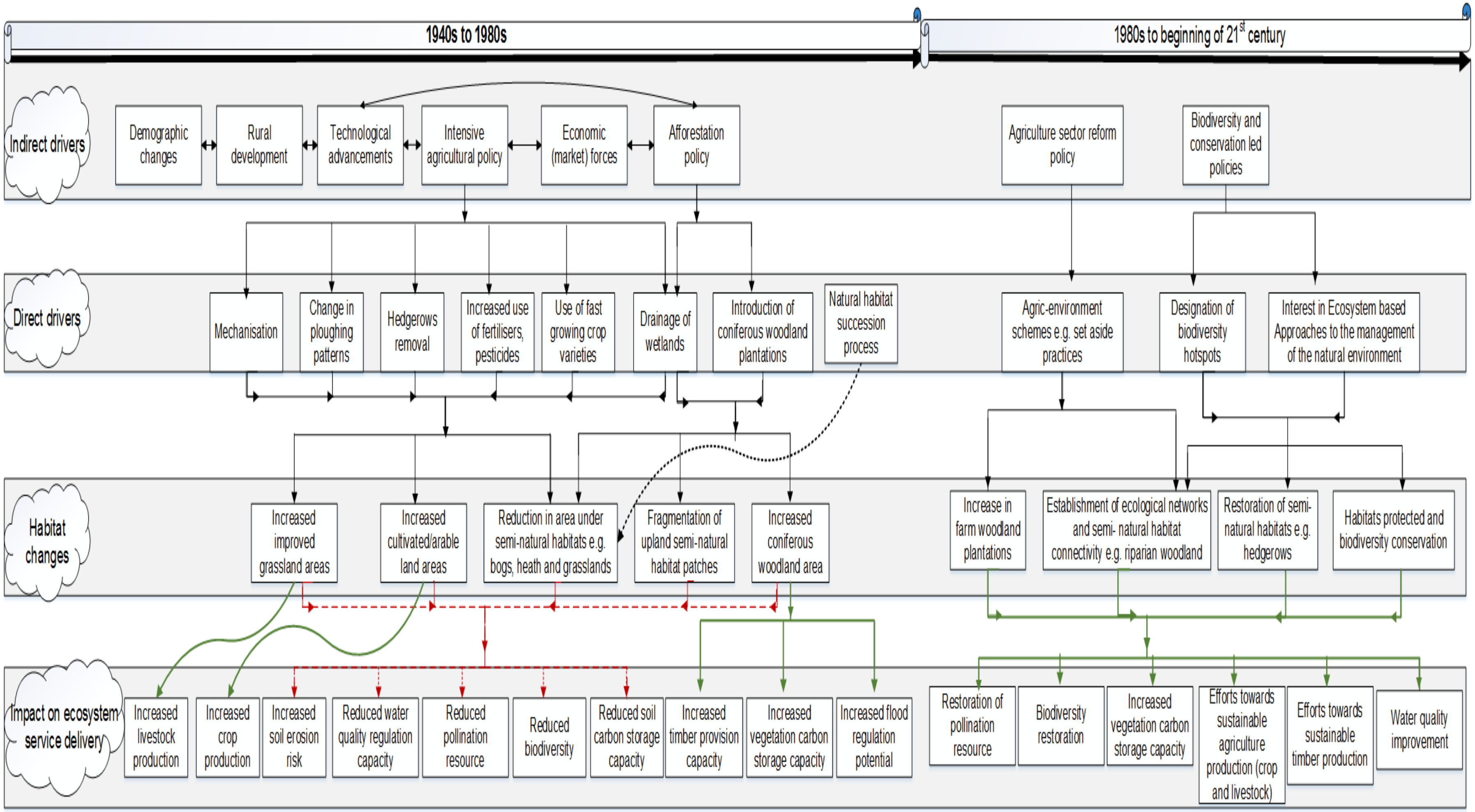

Figure 5-2 Drivers of habitat and ecosystem services changes in the study catchments

Agriculture intensification legislation and policies

Significant changes to the natural environment in the UK, as elsewhere in Europe occurred after the Second World War following the onset of the national reconstruction phase (Bowers, 1985, Shaw, 2007). One of the major aims was to improve food security and self-sufficiency in the UK (Dallimer et al., 2009, Robinson and Sutherland, 2002). To meet this demand, the Agriculture Act of 1947 promoted intensive agriculture to increase livestock and crop production. This led to massive changes in agriculture practices (Cherrill and McClean, 1995, Swetnam, 2007, Thomson et al., 2007) and consequently greatly influenced landscape and habitat changes in the British countryside (Lucas et al., 2011, Blackstock et al., 2007).

The agriculture intensification policy in turn influenced changes in farming systems and farm management in the British countryside, some of which were linked to technological advancements introduced in response to this policy. Such changes as noted by Robinson and Sutherland (2002) included: (a) increased agriculture mechanisation involving the use of tractors for farming instead of the traditional horse draft power. This in turn also influenced other farm management practices including (b) removal of hedgerows to increase field sizes for easier manoeuvring using tractors and a change in ploughing patterns, (c) increased use of inorganic fertilisers, pesticides and fast growing crop varieties to increase production and (d) drainage of wetlands and seasonally flooded grasslands for agricultural expansion.

The entry of the UK into the European Union in 1973 further influenced agriculture activities through the Common Agriculture Policy (CAP) (Bowers, 1985). Some of the CAP incentives for example included favourable prices for livestock production compared to crop production, introducing another dimension of economic drivers. In response to this incentive, some farmers also shifted from mixed farming (crop and livestock production) to increased livestock production (Stevens et al., 2004), as evidenced by the sharp increase in sheep stocking densities after the UK membership into the CAP (Burnside et al., 2003, Dallimer et al., 2009, Cooper et al., 2003).

The influence of the agriculture policy is still evident in the study catchments, as elsewhere in the British countryside. This is partly reflected by findings from this study, which still showed, in 2009, the dominance of intensively managed habitat types such as improved grassland in the low lying areas of the study catchments. Improved grassland was in 1946, and further increased by 2009 to remain, the most extensive habitat type in both catchments, accounting for greater proportions of total catchment areas compared to semi-natural habitat types (Table 4-1). Results from this study show that arable land was also wide spread, in both time periods in the Ale catchment even though it slightly reduced in 2009. In contrast, in the Eddleston catchment, arable land was extensive in 1946 and significantly reduced by 2009. Reasons for such a marked reduction might be linked to market forces such the CAP incentive for increased livestock production. Also, the 1939-1945 war led to the dig for victory campaign-which saw many areas previously not arable become so to feed an island nation in wartime (Harvey and Riley, 2009, Ginn, 2012), so arable higher than might otherwise have been expected very shortly before.

In the context of ecosystem services, as discussed in the previous chapter, the agriculture policy can be interpreted to have favoured and prioritised provisioning ecosystem services while other ecosystem services, notably regulating and supporting were impacted upon. This is because intensively managed habitat types such as arable land are homogenous with less habitat type diversity and hence have low capacities to supply multiple ecosystem services (Burkhard et al., 2012). Likewise, improved grassland is species poor compared to other grasslands such as acid and neutral grassland (Firbank et al., 2013). The increased use of fertilisers, herbicides and pesticides led to farmland bird declines due to habitat and food losses (Donal et al., 2001), while the removal of hedgerows reduced suitable habitat for pollinator species, impacting on both biodiversity and pollination ecosystem services (Maudsley, 2000, Haines-Young et al., 2003). Furthermore, the change in ploughing patterns and removal of hedgerows increased the soil erosion risk and diffuse pollution impacting on water quality ecosystem service (Kay et al., 2012).

Declines in most farmland birds in the study catchments were mainly in the 1970s and 1980s (Taylor and Grant, 2004, Siriwardena et al., 1998). Farmland birds such as Lapwings are noted to have declined over time in the Scottish borders (Foster et al., 2013) due to reduced availability of suitable habitats for such species resulting from loss of hedges and drainage of wetlands. In addition, wading birds such as the Common Snipe, common in wet and boggy areas are also reported to have been affected by drainage of wetlands and agriculture intensification, and these only started showing signs of increase in Scotland in 1994 (Foster et al., 2013).

However, reports from local birdwatchers in the Eddleston catchment show that bird species such as the Blue-tailed, Water Rail and Common Blue Damselfies have started re-apprearing in this catchment (Murray, 2016). The recovery of these bird species is attributed to on going habitat restoration and Natural Flood Management initiatives such as re-meandering of sections of the Eddleston River and creation of ponds.

Afforestation policy

According to Forestry Commission (2015a), it was not until after the Second World War that massive afforestation of coniferous woodland in the uplands of the British countryside was undertaken, with an aim of meeting domestic timber needs in the UK. The mechanical revolution of the period between 1950s and 1970s is noted to have facilitated easy access and afforestation in the uplands (Bunce et al., 2014).

As presented in the results chapter, extensive coniferous woodland plantations in the study catchment`s uplands by 2009 reflect the influence of this policy. Coniferous woodland plantations were non-existent in upland areas of the study catchments in 1946, and only occupied small areas in the low lying areas of these catchments. This habitat type recorded the highest area percentage increase of over 1000% in the Ale catchment and 855% in the Eddleston catchment (Figure 4-10) by 2009. It also spread to occupy significant proportions of total catchment areas, especially the upland areas (Figure 4-12), accounting for 21% (about 3600 ha) of the total Ale catchment area and about 13% (1000ha) of the Eddleston total catchment area.

Corresponding to this was a marked decrease in the area under semi-natural habitat types such as wet bogs, heathland and acid grassland mosaics in the uplands of both catchments by 2009. These were mostly spatially displaced or interspersed with expansive coniferous woodland plantations (Figure 4-4). In so doing, the spread of coniferous woodland plantations in the study catchments` upland areas led to fragmentation of semi-natural habitat patches into smaller sizes by 2009. Such changes in semi-natural habitat patterns in the study catchments were indicated by the reduction in their mean habitat patch size, increase in the number of habitat patches and decrease in the proximity index by 2009.

Fragmentation and degradation of semi-natural and natural habitats have been identified as major factors contributing to biodiversity loss (Haines-Young et al., 2000, Fahrig, 2003, Bennett and Saunders, 2010, Hooftman and Bullock, 2012) especially through the maximisation of provisioning ecosystem services such as food or timber (Dallimer et al., 2009, Millennium Ecosystem Assessment, 2005, Robinson and Sutherland, 2002, Tscharntk et al., 2005). This consequently impacts on multiple ecosystem service delivery as biodiversity underlies key ecological processes and functions and plays a key role in ecosystem service provision (Vermaat et al., 2015). Although the computed landscape metrics – indicating habitat fragmentation and diversity, do not provide any information on species composition or species richness, identified changes to these in this study are presumed to have impacted on biodiversity and hence on multiple ecosystem service delivery in the study catchments.

Land use diversification, biodiversity conservation and environmental management led policies

In the UK, the majority of intensive agricultural and forestry influenced habitat and landscape changes occurred between the 1940s to the 1980s and slowed down thereafter with increasing awareness on the adverse impacts of these on the natural environment (Forestry Commission, 2015a, UK-NEA, 2011, Posthumus et al., 2010). This led to a policy shift and introduction of legislation and policies that put a greater emphasis on sustainable agricultural practices, alongside biodiversity conservation and environmental management.

In response to increasing environmental concerns, the CAP reforms in 1992 introduced agri-environment schemes aimed at reducing overproduction from agriculture (Robinson and Sutherland, 2002, Firbank et al., 2013), while also promoting biodiversity and nature conservation in intensively managed agriculture areas. Set-aside practices are an example of such agri-environment schemes, in which farmers were financially compensated for not farming small parts of their land (Sotherton, 1998). The 2001 Rural Stewardship Scheme in Scotland is an example of such agri-environment schemes where farmers received voluntary incentives for adopting environmentally friendly farming practices through reducing the use of pesticides, restoring or maintaining hedgerows etc. (Scottish Government, 2006). Such practices were aimed at restoring degraded habitats and establishing semi-natural habitat connectivity and ecological networks. In so doing, these would provide corridors for species movement, provide suitable pollinator habitats, food and increase biodiversity, especially for species such as farm birds, whose abundance is noted to have been severely impacted by intensive agriculture practices (Robinson and Sutherland, 2002, McCracken et al., 2012).

The Scotland Rural Development Programme also addresses the intentions of the CAP aimed at encouraging farmers to manage their land for environmental and biodiversity conservation, including key habitats such as hedgerows, arable fields, wetlands, native woodland, uplands heath and moorland (Austin et al., 2015). Some of the observed positive changes resulting from such practices were noted by the countryside survey 2000 (Haines-Young et al., 2000). This, for example, reported a halt in the rate of hedgerow loss in the 1990s and concluded that the negative trends of habitat loss had slowed down during this period.

Findings from this study support the influence of these initiatives. For example, the 2009 habitat maps show increased presence of coniferous woodland plantations within farms in the low-lying areas of the Ale catchment. Such farms in the lower catchment areas of the Ale did not have such woodland plantations in 1946. By 2009, there was also increased presence of riparian mixed woodland in the study catchments, which was not there in 1946. The presence of these in 2009 could be interpreted as indications of implementation of farm woodland land schemes (Forestry Commission, 2015c), in the low lying areas of the study catchments which are dominated by intensive farming activities. Riparian woodland can mitigate against diffuse pollution while also serving as ecological networks (CJC Consulting, 2002, Naiman et al., 1997).

Parallel to the agriculture sector reform was the increase in biodiversity conservation and environmental management led legislation and policies. The table below gives an overview of such legislation, policies, directives and strategies which focussed on nature conservation and broadly the management of the natural environment, including river catchments. These include the major international conventions and European level directives, as well as UK and Scottish level legislation and policies. These, as elsewhere in Scotland informed catchment and land use management within the Scottish Borders.

Table 5-1: Important International, European, UK and Scotland environmental legislation, policies and strategies

Year	International level	Focus
1971	Convention on wetlands of international importance (Ramsar)	Wetlands protection
1979	Convention on the Conservation of European Wildlife and Natural Habitats	Ecosystem and species protection
1992	Convention on Biological Biodiversity	Biodiversity protection
2000	European Landscape Convention	Landscape protection
Year	**European level**	**Focus**
1979	Birds Directive	Species protection
1992	Habitats Directive	Habitats protection
2000	Water Framework Directive	Integrated water resources management
2007	Floods Directive	Flooding
2010	EU Biodiversity Strategy 2020	Biodiversity conservation
Year	**UK level**	**Focus**
1981	The Wildlife and Countryside Act	Ecosystem and species protection
1994	The conservation of natural habitats regulations	Habitats protection
1997	Hedgerow regulations	Hedgerow protection and restoration
2008	Climate Change Act	Climate change mitigation
2010	The conservation of habitats and species regulations	Ecosystem and species protection
Year	**Scotland level**	**Focus**
2003	Water Environment and Water Services (Scotland) Act	Water resources management
2004	Nature Conservation (Scotland) Act	Biodiversity conservation
2004	Scottish Biodiversity Strategy	Biodiversity conservation
2006	Scottish Forestry Strategy	Sustainable Forest management
2007	Scottish Rural Development Programme	Sustainable agriculture and biodiversity enhancement
2009	Flood Risk Management (Scotland) Act	Flooding
2009	Climate Change Act	Climate change mitigation
2010	National Ecological Network	Habitat restoration and ecological networks
2011	Land use Strategy	Integrated management of the natural environment
2011	Water Environment (controlled Activities) (Scotland) Regulations	Control of diffuse pollution

Source: Modified from UK NEA (2011)

As shown in table 5-1, some of the first legislation introduced in the UK in the 1980s include the Wildlife and Countryside Act (1981) and this afforded protection of the countryside natural environment including its biodiversity and semi-natural habitats (Scottish Natural Heritage, 2015b). The table also lists other significant conventions and directives which were introduced in the 1990s with further emphasis on nature conservation, including the Convention on Biological Diversity (1992). This advocated for an ecosystem approach to the management of the natural environment, calling upon governments to adopt an integrated approach to the management of land, water and living resources in order to conserve biodiversity and ensure its sustainable use (Convention on Biological Diversity, 1992).

The EC Habitats Directive is another example of a European level directive introduced in 1992. This advocated for the maintenance of biodiversity through restoration of natural habitats and wild species including the protection of habitats and species which are of European importance (Joint Nature Conservation Committee, 2015). Associated with this directive were a number of designations such as Special Areas of Conservation and Special Protection Areas, which together make up the suite of sites known as Natura 2000, aimed at protecting habitats such as wetlands, which are of high biodiversity value.

Many of the landscapes and sites within the Tweed catchment are designated as conservation or protected sites. The Tweed River is designated as a Special Area of Conservation as well as a Site of Special Scientific Interest (SSSI). Other designations in the Tweed catchment include Ramsar sites (Scottish Natural Heritage, 2015a). Most of these designations are due to the diverse habitats found in this catchment, which for example, serve as important breeding sites for birds (Tweed Forum, 2010).

Although the aforementioned legislation and policies (Table 5-1) were aimed at protecting sites, specific habitat types and species and identified environmental issues, it can be argued that, they all implicitly relate to ecosystem services. Taken together, they can be viewed as efforts towards restoring multifunctional landscapes, including some of the regulating, supporting and cultural ecosystem services and balancing these with sustainable provision of supply of crop, livestock and timber ecosystem services.

Since the beginning of the 21st century, the Water Framework Directive (WFD) (2000) which in Scotland was translated into the Water Environment and Water Services Act (2003) has a key influence in catchment management. This directive requires member states to implement river basin management plans in order to protect and improve the ecological status of surface and ground waters. It thus provided a framework for addressing challenges and pressures in catchments emanating from flooding, diffuse pollution etc. In line with the requirements of the WFD, SEPA adopted the use of catchments as units for management to allow for integration of different land and water uses within water bodies in Scotland. In its second round of river basin management planning, SEPA went further and characterised all water bodies using an ecosystem services based classification (Scottish Environment Protection Agency, 2016). Such an approach goes beyond understanding the ecological status of water bodies to include understanding linkages and interactions between different land uses within catchments and the multiple ecosystem services provided by these (Blackstock et al., 2015). There are also a number of laws which have influenced catchment management such as the

Flood Risk Management (Scotland) Act (2009), which advocates for sustainable flood management at catchment scales (Spray et al., 2010).

In this regard, the early phase of 21^{st} century continues to place more emphasis on the need to adopt a holistic and integrated ecosystem based approach to the management of the natural environment (DEFRA, 2010, Stoate et al., 2009). The UK National Ecosystem Assessment (2011), following the Millennium Ecosystem Assessment in 2005 also explicitly refers to the ecosystem services concept as a possible framework for assessment and management of the natural environment. This has been followed by lots of research and interest on understanding how this concept can be implemented (Mulder et al., 2015). For example, The Scottish Government recently published the revised Land Use Strategy for 2016-2021, which seeks to utilise an ecosystems approach towards a strategy for land use (Scottish Government, 2016). This strategy also highlights the importance of using catchments as units for environmental management as was done in the LUS pilot project in the Scottish Borders.

Findings from this study can be viewed as supporting what authors such as Dearing et al. (2012) refer to as the “Perfect Storm” metaphor, in which sociopolitical factors influenced the change of the 1946 catchment landscapes from multifuctional landscapes with a potential for sustainable multiple ecosystem service provision into unstable catchment landscapes by 2009 with a limited potential to provide multiple ecosystem services. However, the change and shift in environmental legislation and policies including shifts in agriculture practices towards agri-environment schemes after the 1980s can be seen as efforts towards restoring the sustainability of catchment landscapes through stabilising and reducing the loss of regulating and supporting ecosystem services. Dawson et al. (2010) notes that resilience, stability, durability and robustness are necessary properties for sustainability of socioecological systems such as catchments.This implies that the management of current catchment landscapes needs to be based on understanding of these properties for sustainable multiple ecosystem service provision over long term.

Development

Although there were slight (statistically insignificant), increases in built land in the study catchments, there were however, infrastructure developments such as the introduction of open cast mining and the millennium farm in the Eddleston catchment, which replaced semi-natural habitats in those areas. Such developments reduce the capacity of such areas to supply regulating and supporting ecosystem services. In addition, demographic changes related to ageing and declining rural population increased movements to village towns and urban areas (Miller et al., 2009). The expansion of Peebles village on the floodplains of the Eddleston River for housing for example, impacted on the flood control potential in this catchment, increasing flood risk to properties in this village (Murray, 2015).

Climate change

According to the UK NEA (2011), climate change was not a major influencing factor in habitat and ecosystem service changes between the two time points that this study focusses on in the UK. It is, however, expected to influence such changes in future as the UK Climate Projections (2009) (UKCP09), suggest that Scotland will have warmer and wetter winters while summers will be hotter and drier and frequent heavy rains (Meteorological Office, 2014). This could present more flooding challenges, impacting on runoff, natural flood management capabilities and water quality in the study catchments.

5.3.1 Influence of drivers of change on ecosystem service relationships

Bennett et al. (2009) identified mechanisms influencing relationships in ecosystem services as resulting from: (1) the effect of common drivers and (2) interactions among ecosystem services. Figure 5-3 illustrates how the main drivers of change, discussed in the previous section, and ecosystem service interactions influenced changes in ecosystem service relationships in the study catchments.

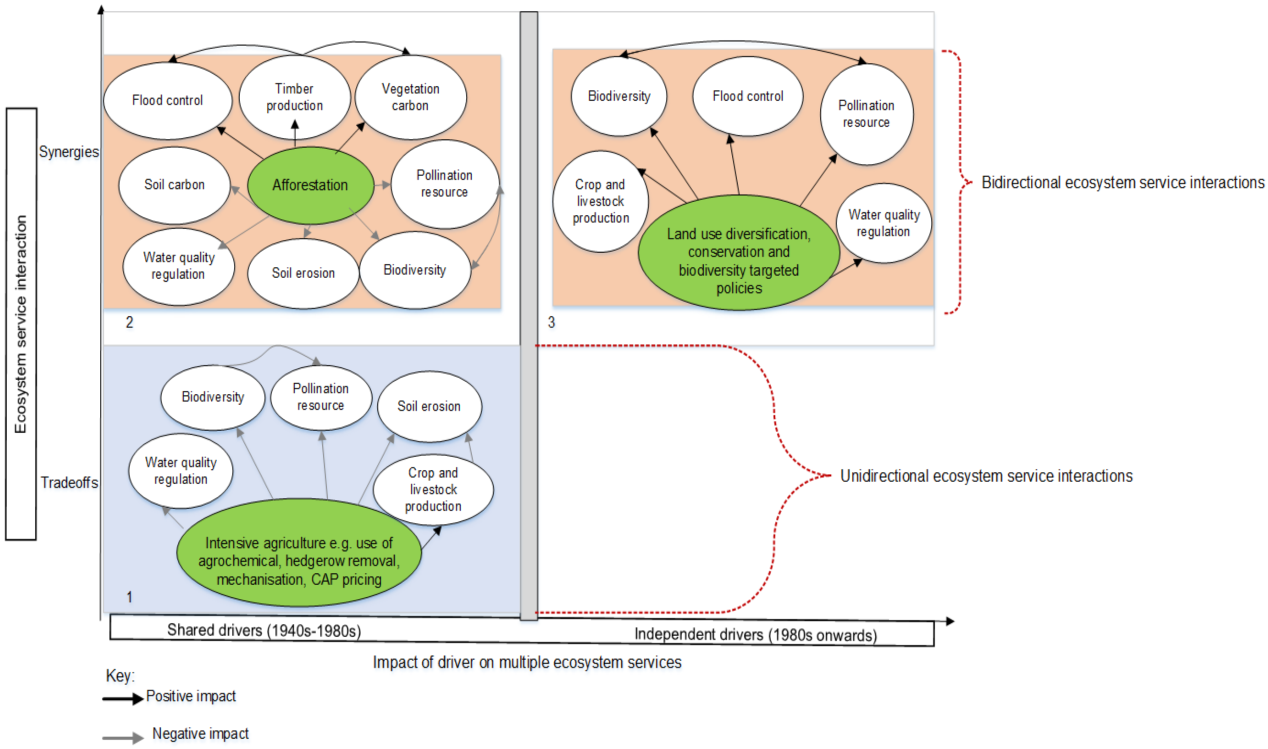

Figure 5-3: Identified ecosystem service relationships
Source: Modified from Bennett et al. (2009)

As illustrated in figure 5-3, ecosystem service relationships can result from shared or independent drivers of change (x axis). The influence of such drivers of change can either be positive or negative in ecosystem service delivery. Section 1 (bottom left), shows that shared drivers of change responding to agriculture intensification policy influenced changes in multiple ecosystem services. However, although this driver had a positive impact on livestock and crop production, it negatively impacted on biodiversity, pollination, water quality ecosystem services. Likewise, the afforestation of coniferous woodland in the uplands positively impacted on timber, flood control and vegetation carbon storage ecosystem services while negatively impacting on other regulating ecosystem services (section 2). Policy shifts in the 1980s onwards, on the other hand can be seen as implicit initiation towards positive impact on multiple ecosystem service delivery (section 3), although such factors, as independent drivers were targeted at specific environmental issues, specific sites, habitat types etc.

Figure 5-3 also shows that ecosystem service relationships also result from interactions among ecosystem services (y axis). Ecosystem service interactions depend on how the level of provision of an ecosystem services impacts (positively or negatively) on the provision of other ecosystem services. Such interactions, as identified by Bennett et al. (2009), can either be unidirectional or bidirectional. Unidirectional interactions occur when the level of provision of an ecosystem services affects the level of provision of an(other) ecosystem services but not vice versa. This often leads to ecosystem service trade-offs. For example, livestock production can impact on water quality ecosystem service but water quality does not in turn impact on the livestock production ecosystem service. Bidirectional interactions occur when the level of provision of an ecosystem service affects and is in turn also affected by the level of provision of other ecosystem services. For example, biodiversity ecosystem service influences pollination ecosystem services and similarly, a change in pollination ecosystem services impacts on biodiversity ecosystem service.

The shifts in policies after the 1980s, can be understood to have influenced alterations in ecosystem service trade-offs, synergies and ecosystem service bundles in the study catchments. For example, the introduction of land use diversification and conservation focussed policies altered the trade-offs in ecosystem services from prioritisation of provisioning ecosystem services towards balancing these with other regulating and supporting ecosystem services. The increased presence of riparian and farm woodland observed in the intensively agriculture managed catchment areas can be viewed as altering ecosystem service relationships towards synergies, ecosystem service bundles and positive bidirectional interactions. This means drivers of change influence changes in ecosystem service interactions and they also alter such ecosystem service interactions over time. Identifying and understanding such drivers of change and their influence on ecosystem service interactions can help to manage trade-offs and synergies in ecosystem services and also help to inform catchment management.

5.4 Implications for catchment management

Habitat changes identified in this study are well recognised and broadly follow those reported elsewhere in the British countryside e.g. Mackey et al. (1994) in Scotland, Hooftman and Bullock (2012) in England, Cooper et al. (2003) in Northern Ireland. However, translating and understanding implications of these habitat changes within the context of ecosystem services brings to the fore their impact on well recognised provisioning ecosystem services as well as less obvious, often less valued regulating and supporting ecosystem services. It has been shown that: (1) a change in one habitat type leads to multiple changes in ecosystem services, (2) changes in the spatial configuration of habitat patches results in spatial dispersion of high ecosystem service supply areas, and (3) interactions of ecosystem services over time showed indications of trade-offs and synergies within and between the study catchments.

Given the current focus and considerations on how the ecosystem based approaches such as the ecosystem service concept can inform catchment management (Wallis et al., 2011), the historic understanding from this study and the influence of spatio-temporal habitat changes on ecosystem service delivery adds an important layer in catchment management. This could particularly be relevant in the implementation of directives and laws such as the WFD or Flood Risk Management (Scotland) Act (2009), which currently have a key influence in catchment management. Many commentators (Blackstock et al., 2015, Brauman et al., 2014, Wallis et al., 2011) acknowledge that the adoption of the ecosystem services approach can add value and assist in the implementation of the WFD – which largely follows the IWRM approach. They further explain that the conceptualisation of the link between the environment and how ecosystem services from the water environment are valued can go beyond the WFD goal of good ecological status to include the suite of ecosystem services provided by catchments including those valued by stakeholders (Blackstock et al., 2015).

Spatial assessment of ecosystem services and understanding changes to these, as illustrated in this study, can assist in ecosystem restoration efforts. For example, catchment areas that had a high potential for multiple ecosystem service provision in the past can be identified and targeted for management action, as illustrated in Figure 5-4 below. Also, remaining semi- natural habitat patches, important for multiple ecosystem service delivery can be identified and inform prioritisation of such areas for conservation efforts. In managing such habitat types understanding thresholds in habitat sizes and spatial location on ecosystem service delivery can add value in catchment management.

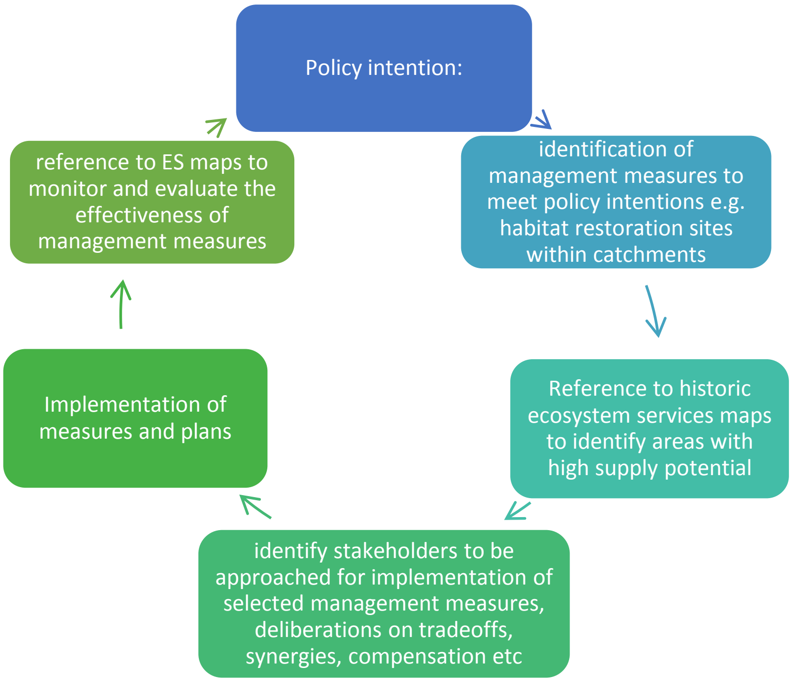

Figure 5-4: Illustration on the potential use of a historic ecosystem service baselines in catchment management

As illustrated in the figure above, information on past ecosystem services changes can be used as a baseline to guide management intentions, evaluate effectiveness of policy implementation and against which change can be monitored. On this basis, it can be argued that the use of historic baselines to inform future management of ecosystem services adds an important strand in catchment management. Most studies e.g. Raudsepp-Hearne et al. (2010) and Maes et al. (2012b) have focussed on contemporary delivery of different ecosystem services and used the present/current as a baseline to inform future ecosystem services delivery and management, with limited focus on past ecosystem services and how they have changed over time. Yet, as argued by Johansson et al. (2008), explaining present ecosystems without historic understanding provides a limited perspective on causes of ecosystem changes.

Understanding past ecosystem services can also help explain ongoing ecosystem changes and help put the current situation into context (Swetnam et al., 1999). This is particularly applicable to river catchments, the current state of which can be attributed to past modifications and changes. For example, the mid-20th century is of importance in catchment management as the period that led to massive alteration of regulation functions of catchment landscapes (Newson, 1977). Additionally, it is also the period that marked the onset of transformation of traditional catchment landscapes in the British countryside as in most parts of Europe, to modern, intensively managed catchment landscapes (Tscharntk et al., 2005). In this regard, recognising changes in ecosystem services can deepen understanding on how catchments can be managed in future.

Stakeholder engagement is another key aspect of the ecosystem services approach (Blackstock et al., 2015) as well as the central focus of most current directives influencing catchment management such the WFD. Presenting local stakeholders with ecosystem services maps of the past as well as the present, such as the ones produced in this study, could stimulate discussions, questions and local narration of changes and in so doing help capture local knowledge. Local stakeholders have been observed to be crucial in providing traditional knowledge which can influence how ecosystems are managed (Raymond et al., 2010).

Local stakeholder discussions and deliberations on ecosystem services would also ensure that local preferences, perceptions, values and interests are known even in cases that entail trade-offs. Involvement of relevant stakeholders can further assist in the prioritisation of valued ecosystem services (Brauman et al., 2014). In so doing, local stakeholders e.g. land owners and land managers as local actors influencing habitat and ecosystem service changes, can further inform catchment management. This is particularly in view of concerns and queries over meaningful local stakeholder involvement in environmental decision making and policy implementation e.g. as levelled against the implementation of the WFD in the EU (Niasse and Cherlet, 2015). Similarly, the Flood Risk Management (Scotland) Act (2009), which although intended to take an integrated catchment approach, is noted to give little recognition to local stakeholder engagement (Spray et al., 2010).

5.5 Chapter summary

Two main phases of policy shifts which have influenced changes in habitats and ecosystem services in the study catchments are identified. The influence of agriculture intensification and afforestation legislation and policies (1940s to 1980s) marked the first phase. These, together with other drivers of change in turn influenced a number of simultaneous changes in farming systems, farmer behaviour and technology e.g. mechanisation of farming, increased use of fertilisers and pesticides etc. The second phase (from the 1980s to the beginning of the 21st century) was marked by policy shifts, including the reform of the agriculture sector and introduction of land use diversification, biodiversity and environmental management focussed laws and policies.

The identified drivers of change in turn influenced ecosystem service interactions resulting in ecosystem service trade-offs and synergies over time. For example, provisioning ecosystem services were prioritised and maximised during the intensification phase, while regulating and supporting ecosystem services were traded off. The shift in these legislation and policies over time towards restoration of regulating and supporting ecosystem services also influenced alterations in ecosystem service interactions towards ecosystem service synergies and bundles.

The understanding on influence of spatio-temporal habitat changes on ecosystem service delivery is argued to be an important strand emerging from this study which could further inform catchment management. It is suggested that planning for future ecosystem services in catchment management should be based on historic baselines as these could help explain the current catchment landscapes. In so doing this would also inform and complement implementation of a number of policies and directives such as the WFD, which have influenced catchment management in the recent past. The historic focus of this study has also provided the evidence base on past ecosystem services, addressing assumptions that have been made about these.

6 Conclusion and Recommendations

6.1 Chapter introduction

This study sought to assess and understand how habitat changes across space and time influence changes in ecosystem services delivery at a local catchment scale, and how such an understanding could potentially inform catchment management using the ecosystem services concept. To do this, this study utilises current proxy based approaches of using habitat maps to map past ecosystem services. These were in turn compared with pre-existing contemporary ecosystem service maps to assess spatio-temporal changes to these over time. An understanding on how past habitat changes and modifications have influenced ecosystem service delivery over time is a key contribution from this study. Key conclusions from this study show that ecosystem service delivery is not only affected by changes in gross area of constituent habitats, but also by spatial changes in the configuration and distribution of these habitats.

6.2 Synthesis of thesis chapters

The Introduction chapter (1), presents the background to this study, highlighting increasing concerns, worldwide, over unsustainable exploitation, pollution and severe degradation of river catchments as important ecosystems for human survival. This damage is associated with adverse impacts of human activities on these ecosystems which, although crucial for human survival are among the most threatened. The chapter also explains how increased awareness and recognition of human impacts have drawn increased interest in both science and policy, towards understanding human-nature relations and how ecosystem based approaches like the ecosystem services concept can inform sustainable management of the natural environment.

The Literature review (chapter 2), seeks to provide a general understanding of the ecosystem services concept, its origins, classification and current debates around the ecosystem services concept. The chapter also discusses the increasing importance attached to catchments as units for environmental management and how notions such as the ecosystem service concept can inform their management. The literature review chapter also notes that mapping ecosystem services is at the core of contemporary approaches to assessing ecosystem services. Mapping methods and approaches developed to this effect are reviewed. The need to understand changes in ecosystem services resulting from habitat modifications over time was identified as a key knowledge gap, as limited studies have attempted to map and assess changes to past ecosystem services (Haines-Young et al., 2012). In response to this knowledge gap, the following research questions were posed in this study.

- What is the historic state and patterns of habitat change in the Ale and Eddleston catchments?
- What are the historic ecosystem services in these catchments and what changes have occurred to these?
- How have the identified changes in habitats influenced ecosystem service delivery in both catchments?
- What are the similarities and differences between habitat change and ecosystem service delivery across both catchments?
- Which key factors influenced habitat and ecosystem service changes in the study catchments?

The Methodology chapter (3) presents the data collection and processing procedures devised to map habitats and to enable assessment of spatio-temporal changes in habitats and ecosystem services delivery. Two study catchments were selected to contextualise this study and the ecosystem services cascade was used as an underlying conceptual framework. It was however, modified in line with the aims of this study to map and assess changes in habitats and ecosystem service delivery.

In temporal terms, the baseline for this study was derived from mid-20th century air photography coverage of both catchments flown by the RAF. While this photography was not originally intended for this research purpose it was possible nevertheless to construct digital photo mosaics of the landscapes of the study catchments. Using GIS and other ancillary data sets, the photo mosaics were in turn used to derive the historic (1946) habitat maps for the study catchments. The SENCE ecosystem services mapping approach, informed by expert opinion and scientific literature informed look up tables and other spatial data, was then used to translate generated habitat maps into ecosystem service supply maps. A comparison of the generated historic habitat and ecosystem services maps with the current (2009) maps for the study catchments provided the basis for assessing spatio-temporal changes both in habitats present in, and ecosystem services delivered by, either catchments. The assessment was informed by input from local stakeholders regarding contemporary priority ecosystem services (Spray, 2014). These were timber provision, livestock production, crop production, pollination, biodiversity, soil carbon storage, vegetation carbon storage, land erosion risk, water quality regulation and flood regulation.

Findings from the assessment are presented in the results chapter (4). Findings show that both study catchments recorded an increase in intensively managed habitat types by 2009, notably in the area under improved grassland and coniferous woodland plantations. Improved grassland was extensive in the low-lying catchments areas while coniferous woodland plantations were mainly found in the uplands. In contrast, a decrease in area under arable land is also observed in both catchments. The aforementioned changes were also coupled with a decrease in both catchments in the area under semi-natural habitat types, especially bogs, heathland, hedgerows and unimproved grassland areas.

Linked to this was a change in spatial habitat patterns and configuration within the catchment landscapes, including fragmentation of semi-natural habitat patches into smaller sizes. Semi-natural habitats were mostly converted or interspersed with coniferous woodland plantations, especially in the uplands. Conversely, intensively managed habitat types such as improved grassland did not show similar evidence of spatial fragmentation, having a higher proximity index and larger mean patch sizes in 2009. Consequently, both catchment landscapes had become less diverse and more homogenous in 2009 compared to 1946.

Results from analysis of changes in ecosystem services delivered within both catchments, show on the one hand - that both catchments had an increased capacity in 2009 to supply livestock production, timber production, vegetation carbon storage and flood regulation ecosystem services whereas on the other hand, their capacity to supply biodiversity, pollination, water quality and soil carbon ecosystem services was reduced.

Interpretation of these findings within the ecosystem services context, as discussed in chapter 5, showed that: firstly, a change in one semi-natural habitat type leads to multiple changes in ecosystem services, confirming the role and importance of semi-natural habitats in multiple ecosystem service delivery as also, for example observed by Burkhard et al. (2012). Also, spatial changes in the configuration of semi-natural habitat patches led to spatial dispersion of high ecosystem service supply areas, leading to decrease of high capacity areas for the supply of regulating and supporting ecosystem services.

Findings also indicate trade-offs in ecosystem services as provisioning ecosystem services were prioritised in the past while regulating and supporting ecosystem services were impacted upon and traded off. This may be attributable to possible lack of knowledge or interest in the value of regulating and supporting ecosystem services in the past. However, the policy shifts from the 1980s can be viewed as altering such trade-offs towards ecosystem service bundles and synergies.

Also emerging from the analysis of ecosystem service relationships were spatial overlaps in some ecosystem service supply areas e.g. timber provision, vegetation carbon storage and flood control potential, suggestive of ecosystem service bundles and synergies. However, while this thesis acknowledges ecosystem service bundles associated with the increase in coniferous woodland plantations, which also somewhat compensated for soil carbon storage ecosystem services, it can be argued that the supply capacities for vegetation carbon and flood control is a short term positive impact, likely to be interrupted by timber harvesting activities.

The broadly similar patterns of habitat and ecosystem service changes in the study catchments may be attributable to similar drivers of change, although the extent of the impact of these factors differed between the study catchments. As reported elsewhere in the Scottish and wider British countryside e.g. Miller et al. (2009) and UK-National Ecosystem Assesment (2011), the main drivers of change are linked to socio-political factors which acted interdependently with other factors to influence these changes. Among these, agriculture intensification and afforestation legislation and policies introduced in the 1940s, after the Second World War were identified as key influencing factors (Bowers, 1985). Alongside with economic, technological advancements factors (Robinson and Sutherland, 2002), these in turn influenced changes in farm management and practices including use of tractors, fertilisers, chemical pesticides resulting in habitat and ecosystem service changes.

Increased awareness of the adverse impact of agriculture intensification led to policy reforms from the 1980s, towards promotion of sustainable agriculture practices (Firbank et al., 2013), through for example, agri-environment schemes. Parallel to this was also increased introduction of legislation and policies aimed at nature conservation, sustainable management of the natural environment and biodiversity conservation (Stoate et al., 2009). While these policies were targeted at specific sites, species and specific environmental issues, taken together, they can be viewed as efforts towards restoring multifunctional landscapes within intensively managed areas. In so doing, it can be argued that these policies implicitly relate to ecosystem services, including some of the regulating, supporting and cultural ecosystem services and balancing these with sustainable provision of supply of crop, livestock and timber ecosystem services. Since the start of the 21st century, there has been more emphasis in both policy and science, on integrated ecosystem based approaches, making explicit reference to the ecosystem services concept as a possible framework for the management of the natural environment (Mulder et al., 2015).

6.3 Key conclusions

This study has demonstrated that understanding habitat changes within the ecosystem service context provides further insights not depicted in habitat change analysis alone. Findings from this study show that translating and understanding habitat changes within the context of ecosystem services brings to the fore their impact on well recognised provisioning ecosystem services as well as less obvious, often less valued ecosystem services. For example, while converting uplands bogs to coniferous woodland was a “single” habitat change, this led to multiple ecosystem service changes (in soil carbon storage, biodiversity, pollination etc.). Based on this, this study concludes that a change in one habitat type leads to multiple changes in ecosystem services, especially regulating and supporting ecosystem services.

This study has also demonstrated that the introduction of intensively managed habitat types led to fragmentation of semi-natural habitats, especially in the uplands of the study catchments. These spatial changes led to the spatial dispersion and decrease of areas with a high capacity to supply multiple ecosystem services. This means that changes in spatial configuration and arrangement of (semi-natural) habitats leads to a decrease in high capacity supply areas for regulating and supporting ecosystem services. On this basis this study concludes that ecosystem service delivery is affected by spatial changes in the distribution and configuration of habitats and not just on their gross area.

The multifunctional role of both study catchment landscapes changed over time into intensively managed ones, with a higher capacity to provide provisioning ecosystem services, both in the low lying and upland areas, while their capacity for multiple ecosystem provision was reduced. However, evidence of increased presence of habitat restoration measures within intensively managed low lying catchment areas e.g. farm woodland and riparian vegetation in 2009 was interpreted as efforts towards restoring multifunctional catchment landscapes.

Identified drivers of change influenced interactions among ecosystem services resulting in ecosystem service trade-offs, synergies and ecosystem service bundles over time. The increase over time in intensively managed habitat types led to prioritisation of provisioning ecosystem services while regulating and supporting ecosystem services were traded-off. By contrast, in addition to timber provision, the increase in coniferous woodland plantations in the uplands increased supply capacities for vegetation carbon and flood regulation ecosystem services, suggestive of ecosystem service bundles and/or synergies in ecosystem service delivery.

Although the study catchments depicted broadly similar patterns of habitat and ecosystem service changes, there were, however differences in the extent of these changes. The Eddleston catchment for example, had a lower Shannon diversity Index compared to the Ale catchment in 2009, implying that it was more simplified and homogenous than the Ale catchment. The Eddleston catchment also recorded an increase in built land due to expansion of Peebles and Eddleston Villages and the introduction of an open cast mining area. While the study catchments also differ in sizes, the differences in the extent of habitat changes can be attributable to varying land use preferences and choices in these catchments over time, as well as land ownership. This is of importance in understanding local stakeholder preferences and values in ecosystem services to inform catchment management.

An understanding on how past habitat modifications and changes have influenced changes in ecosystem services is arguably a valuable contribution emerging from this study. In this thesis, it is argued that recognising and understanding change in ecosystem services is an important strand in catchment management. In this regard, it is suggested that future ecosystem services need to be planned on historic baselines rather than the current which most studies have relied on. In particular, the use of time periods such as the mid-20th century that are known to have led to significant changes in catchment landscapes in countries such as the UK to inform their future management.

6.4 Implications for policy development

Findings from this study have implications for policy development, especially those policies that intend to take an ecosystems approach to catchment management. Examples of legislation and policies, relevant to the Tweed catchment as elsewhere in Scotland include the Water Environment and Water Services (Scotland) Act (2003), Flood Risk Management (Scotland) Act (2009) and the Land Use Strategy (2011). The development of these laws and policies can be informed by:

- The historic information from this study e.g. changes in the spatial location of high ES supply areas in catchments, can add an important layer in the contemporary interaction and opportunities ecosystem services maps produced during the pilot LUS project in the Scottish Borders. This historic change information could be utilised in Phase 2 of the LUS project to inform targeting of catchment areas for restoration measures. For example, catchment areas that had a high potential for multiple ecosystem service delivery can be identified and targeted for enhanced multiple ecosystem service provision. Organisations like the Tweed Forum, Scottish Borders Council and SEPA can make reference to such historic information to identify such sites within catchments and inform trade-offs, conflicts analysis etc. Other organisations such as the Forestry Commission can also identify opportunities for tree planting and multiple ecosystem service provision, including diffuse pollution control as part of the LUS Framework.

- The historic ecosystem services maps produced in this study, which to the knowledge of this research have not been produced before for the study catchments, can serve as a communication tool in facilitating discussions about change with local stakeholders. Local stakeholder engagement is the central focus for implementation of environmental policies and directives such as the WFD. Organisations involved in local stakeholder engagement, such as Tweed Forum can present maps of the past as well as the current ones to stimulate discussions among stakeholders. In the process local knowledge on for example, factors influencing the identified habitat and ecosystem service changes as well as local views on how catchment landscapes can be managed for future ecosystem service delivery under different factors such as climate change or different policies can be captured.

- This study could contribute towards linking local level and national level management intentions. For example, the SBC could utilise the historic ecosystem service maps to guide the development of Local Biodiversity Plans, in line with the Scottish National Strategy and the EU Biodiversity Strategy for 2020, which plans to map and assess the state of past ecosystem services in the member states. In so doing this can answer policy questions related to state and trends in ecosystem service provision over time.

6.5 Recommendations for future work

Areas for future research identified in this study include the following:

The first identified area for future work is the need to understand local stakeholder perceptions of change and how they value ecosystem services. Due to time constraints, this study could not engage with local stakeholders in the study catchments to capture their understanding and narration of the observed habitat and ecosystem service changes, as well as how local stakeholders perceive the future management of these catchments under different factors. In so doing, this could also capture changes in cultural ecosystem services, which were excluded in this study. Socio-economic aspects are an important strand of the ecosystem services concept, as this approach is a bridge between biophysical and social sciences. This means a comprehensive understanding of this concept needs to capture these dimensions to inform environmental decision making and policy implementation.

Secondly, further work can also complement this study by understanding how identified changes in the ecosystem service supply areas in the study catchments corresponded with changes in ecosystem services flows and demand areas. These dynamics in ecosystem services are increasingly considered important in understanding the sink areas for ecosystem services and is seen important in informing emerging notions like payments for ecosystem services.

Thirdly, future work can focus on understanding, in detail interactions among different ecosystem services, including ecosystem service bundles, trade-offs and synergies in ecosystem services delivery over time as this study relied on relative approaches to assessing these. Linked to this is the need to also understand functional relationships between habitats and ecosystem service delivery as contemporary approaches rely on simplifying approaches. For example, there are uncertainties and questions related to linearity or otherwise in any loss of ecosystem services as habitats deteriorate (in quantity, quality or location), ecosystem service tipping points, which future work can focus on.

Fourth, future work can be extended to other sub catchments of the Tweed catchment or other catchments elsewhere to compare changes and establish similar historic baselines on the state of ecosystem services. This would help understand whether patterns of change are different or similar and inform catchment management. Such an understanding would also be important in informing policies such the Land Use Strategy (2011).

Lastly, future work could also map and asses the state of habitats and ecosystem services in the study catchments around the 1980s, before the onset of land use diversification and conservation led policies. This could enable assessing the effectiveness of these policies as in study catchments as the analysis of habitat and ecosystem service changes in this study was based on two time points of 1946 and 2009. These two time points are arguably apt as they capture, in 1946, the onset of major factors which resulted in substantial habitat and ecosystem service changes in Scottish catchment landscapes. The 2009 snapshot shows the state of these catchment landscapes 60 years later, at a time of change towards land use diversification and biodiversity conservation led policies and increased awareness on multi-benefits provided by catchment landscapes. However, the produced static 1946 and 2009 ecosystem service maps do not capture temporal changes that occurred to habitats and ecosystem services between these two dates and future work can focus on this.

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Appendices

Appendix 1: Legislation and Policies influencing management of freshwater environment in Scotland

- Climate change Act (2009) and Climate change adaptation framework: this focuses on adaptations to the changing climate and measures that have to be implemented (Scottish Government, 2013).
- Flood Risk Management Act (2009): this is aimed at integrated and coordinated process of managing flood risk at both national and local/catchment levels.
- Water Environment (Controlled Activities) (Scotland) Regulations 2011: Diffuse pollution is regulated under these regulations in which activities such as abstractions, impoundments and engineering works are controlled.
- Scottish Rural Development Programme (2007-2013): this focuses on economic, environmental and social measures in order to support and strengthen rural communities in local innovation. It brings together a wide range of formerly separate support schemes including those covering the farming, forestry and primary processing sectors, rural enterprise and business development, diversification and rural tourism (Scottish Government, 2013).
- Land Use Strategy (2011): Its objectives are based on the principles of sustainability relating to economy, environment and communities and the need for an integrated approach towards the management of the natural environment (Scottish Government, 2013).
- Scottish Forestry Strategy (2006): this mainly focusses on sustainable forest management through for example increasing woodland cover through afforestation initiatives.
- National Ecological Network (2010): this is aimed at habitat restoration projects through for example wetland restoration projects (Scottish Government, 2013).
- Scottish Biodiversity Strategy (2004): aimed at biodiversity conservation. This is to be achieved through activities such as biodiversity restoration and engagement of the public in biodiversity planning and management.

Appendix 2: Principles of the Ecosystem Approach

1. The objectives of management of land, water and living resources are a matter of societal choice
2. Management should be decentralised to the lowest appropriate level
3. Ecosystem managers should consider the effects (actual or potential) of their activities on adjacent and other ecosystems
4. Recognising potential gains from management, there is usually a need to understand and manage the ecosystem in an economic context. Any such ecosystem management programme should:
   - Reduce those market distortions that adversely affect biological diversity;
   - Align incentives to promote biodiversity conservation and sustainable use; and
   - Internalise costs and benefits in the given ecosystem to the extent feasible.
5. Conservation of ecosystem structure and functioning, in order to maintain ecosystem services, should be a priority target of the ecosystem approach.
6. Ecosystem must be managed within the limits of their functioning
7. The ecosystem approach should be undertaken at the appropriate spatial and temporal scales
8. Recognising the varying temporal scales and lag effects that characterise ecosystem processes, objectives for ecosystem management should be set for the long term
9. Management must recognise that change is inevitable
10. The ecosystem approach should seek the appropriate balance between, and integration of, conservation and use of biological diversity
11. The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices
12. The ecosystem approach should involve all relevant sectors of society and scientific disciplines.

Source: www.cbd.int/ecosystem/principles.shtml

Appendix 3: Eddleston sortie plots

Eddleston Sortie Plots purchased from the RCAHMS

Appendix 4: Phase 1 Habitat classes

A. Woodland and scrub													
A1. Woodland						**A2. Scrub**		**A3. Parkland/scattered trees**			**A4. Recently felled woodland**		
Broadleaved		Coniferous		Mixed		Dense/continuous	Scattered	Broadleaved	Coniferous	Mixed	Broadleaved	Coniferous	Mixed
Semi-natural	Plantation	Semi-natural	Plantation	Semi-natural	Plantation								
B. Grassland and Marsh													
B1. Acid grassland						**B2. Neutral grassland**		**B3. Calcareous grassland**			**B4. Improved grassland**	**B5. Marsh/marshy grassland**	**B6. Poor semi-improved grassland**
Unimproved		Semi-improved				Unimproved	Semi-improved	Unimproved	Semi-improved				
C. Tall herb and fern													
C1. Bracken						**C2. Upland species-rich ledges**		**C3. Other**					
Continuous		Scattered						C3.1 Tall ruderal	C3.2Non-ruderal				
D. Heathland													
D1. Dry dwarf shrub heath						**D2. Wet dwarf shrub heath**		**D3. Lichen/bryophte heath**			**D4. Montane heath/dwarf herb**	**D5. Dry heath/acid grassland mosaic**	**D6. Wet heath/acid grassland mosaic**
E. Mire													
E1. Bog					**E2. Flush and spring**			**E3. Fen**			**E4. Bare peat**		
E1.6.1 Blanket bog	E1.6.2 Raised bog	E1.7. Wet modified bog	E1.8 Dry modified bog		E2.1 Acid/neutral flush	E2.2 Basic flush	E2.3 Bryophyte dominated spring	E3.1 Valley mire	E3.2 Basin mire	E3.3 Floodplain mire			
F. Swamp, marginal and inundation													
F1. Swamp						**F2. Marginal and inundation**							
						F2.1 Marginal vegetation	F2.2 Inundation vegetation						
G. Open water													
G1. Standing water						**G2. Running water**							
H. Coastland													
H1. Intertidal	**H2. Saltmarsh**			**H3. Shingle above high tide mark**	**H4. Boulders/rocks above high tide mark**	**H5. Strandline vegetation**	**H6. Sand dune**			**H8. Maritime cliff and slope**			
	H2.3 Saltmarsh/dune interface	H2.4 Scattered plants	H2.6 Dense/continuous				H6.4 Dune slack	H6.5 Dune grassland	H6.6 Dune heath	H8.1 Hard cliff	H8.2 Soft cliff	H8.3 Crevice/ledge vegetation	H8.4 Coastal grassland
							H6.7 Dune scrub	H6.8 Open dune		H8.5 Coastal heathland			
I. Rock exposure and waste													
I1. Natural					**I2. Artificial**								
I1.1 Inland cliff	I1.2 Scree	I1.3 Limestone	I1.4 Other exposure	I1.5 Cave	I2.1 Quarry	I2.2 Spoil	I2.3 Mine	I2.4 refuse-tip					
J. Miscellaneous													
J1. Cultivated/disturbed land				**J2. Boundaries**				**J3. Built-up areas**			**J4. Bare ground**	**J5. Other habitat**	
J1.1 Arable	J1.2 Amenity grassland	J1.3 Emphemeral/short perennial	J1.4 Introduced shrub	J2.1 Intact hedge	J2.2 Defunct hedge	J2.3 Hedgerow with trees	J2.4 Species rich hedges	J3.4 Caravan site	J3.5 Sea wall	J3.6 Buildings			
				J2.5 Wall	J2.6 Ditch	J2.7 Boundary removed	J2.8 Earth bank						

Appendix 5: Ancillary datasets used in air photo interpretation.

Dataset layer	Use	Format	Source*
1946 orthophoto/photo mosaic *(Scale 1: 10 000)*	Base map that facilitated the editing of the current habitat map	Raster (Black and white)	Stage 1 of data collection in this study
Habitat map - current (2009)	Top most layer which was edited to reconstruct the 1946 habitat map	Vector (shape file)	Scottish Borders Council/Environment Systems
Current aerial photography *(25cm spatial resolution and suitable for display at 1: 2500 scale)*	Reference layer used for cross checking, identifying and viewing habitat features in colour	Raster (colour i.e. Red, Green and Blue), 1km x 1km tiles (tiff format)	www.getmapping.co.uk Land map/Environment Systems
DTM Hill shade *(Scale 1: 10 000)*	Reference layer used to assign a habitat class based on its location within the landscape and differentiate between upland and lowland areas	Raster (5km x 5km tiles)	Ordnance Survey /Environment Systems
Soil map	Reference layer used to check underlying soil type especially peat soils.	Vector (shape file)	Environment Systems /James Hutton Institute

Dataset layer	**Use**	**Format**	**Source***
Topographic maps (OS) *(1: 10 000 and 1: 25 000)*	Reference layer used to identify place and river names	Raster (tiff), 10 km x 10km tiles	Ordnance Survey
Hedgerows layer	Used for comparison to identify places that still have hedgerows from the past.	Vector (shape file)	Scottish Borders Council / Environment Systems
Sub catchment boundaries shape files	Used to guide the interpretation so that it's confined within the study area boundaries.	Vector (shape file)	Scottish Borders Council / Tweed Forum

*Copy right permission was granted by the respective suppliers of these data sets

Appendix 6: Datasets and attributes used to map the selected historic ecosystem services

Agricultural crops			NEA service type: Provisioning
This map covers areas used for crop production, the intensive production of arable crops and in some cases the small scale vegetable production in the backyard gardens/allotments.			
Significant effects	Data used	Example attributes	Indicative scoring
Likelihood of land cover to support food production	Phase 1 (1946) habitat layer	Arable Not arable	High Low
Agricultural livestock			NEA service type: Provisioning
This map covers areas which support livestock including arable crops grown for animal feed, intensively grazed areas and extensive permanent grazing regimes. Assumed that improved and semi improved grasslands are provide this ES.			
Significant effects	Data used	Example attributes	Indicative scoring
Presence of suitable grazing environments	Phase 1 habitat layer (1946)	Improved grassland Semi-improved grassland/other habitat mosaics	Medium Very low
Timber resource			NEA Service type: Provisioning
The map covers areas within the land that have woodland plantations and semi natural woodland occurrences. Since plantation woodland has management and growth stages, the type of woodland and planting regime affect how long until the timber resource is ready. Late stage forestry, mature coniferous plantations were given the highest score as they are most likely to provide the maximum timber resource. Recently planted and felled woodland was given a lower score as it will take years before timber is available from such sites. Broadleaved and mixed woodland were given a very low score as few trees are felled at a time for specific site management purposes.			
Significant effects	Data used	Example attributes	Indicative scoring
Provision of coniferous plantation	Phase 1 habitat layer (1946)	Plantations Other woodlands	Productive Non-productive
Soil carbon storage			NEA service type: Regulating
Soil carbon storage results from interactions of different ecological processes. The amount of organic matter present within the soil profile is an important component which contributes to this ecosystem service. Soil organic matter is a heterogeneous mixture of organic compounds that are highly enriched in carbon, ranging in decomposition from leaf litter, to highly decomposed material (humus). Soil organic carbon levels of different soil types are directly related to the amount of organic matter contained in soil from growth and death of plant roots and foliage, as well as indirectly from the transfer of carbon-enriched compounds from roots to soil microbes. Inorganic carbon is not readily released to the atmosphere or water from the soil so it has not been considered in this analysis.			
Significant effects	Data used	Example attributes	Indicative scoring
Presence of organic carbon in the soil	Soils National Soil Survey of Scotland 1:250,000 (including SNH	Organic soils Mineral soils	high Low

	soil carbon classification)		
Topography suitable for soil carbon accruement	Elevation Slopes derived from DTM	Shallow slope Steep slopes	high low
Vegetation cycle accrues / releases soil carbon	Phase 1 habitat layer (1946)	Wetlands and woodlands Heathland Semi-natural grassland Improved grassland High intensity agriculture	high medium low very low negative
Vegetation carbon storage			NEA service type: Regulating
Atmospheric carbon is sequestrated by, and stored in, vegetation. Habitat type is a key determinant of vegetation carbon storage, the more biomass that is present in the vegetation layer the more carbon is stored, with mature woodland at one end of the spectrum and grasslands at the other end. It has been estimated that woodlands and forest vegetation hold up to 80% of the UK total vegetation carbon with those habitats managed for arable and horticultural crops storing the least carbon in their vegetation.			
Significant effects	Data used	Example attributes	Indicative scoring
Biomass presence	Habitats Phase 1 habitat layer (1946)	Woody species Other scrub vegetation Other short vegetation	high medium low
Water quality regulation			NEA service type: Regulating
Water quality is influenced by both natural processes and human activities. Soil temporarily stores water that falls as rain and subsequently releases it to rivers and streams, or adds it to the overall groundwater resource. Some soil types effectively filter water as it percolates through it, whilst others add to the suspended particulate matter and mineral content of the water. Steep slopes shed water more rapidly than shallow slopes. Habitat type, through its link to vegetation structure and type and soil type, has an important influence on water quality. Some vegetation species play a role in water purification.			
Significant effects	Data used	Example attributes	Indicative scoring
Presence of vegetation	Phase 1 habitat layer	Woodland Hedge Heathland Bog Arable	moderate/high moderate moderate moderate/low low/negative
Filtration effect of the soils	Soils National Soil Inventory Scotland 1:250,000	Brown earths Peaty soils	moderate/high low
Slope is linked to flow rate	Elevation Slopes derived from DTM	Steep slopes	Negative
Land erosion risk			NEA service type: Regulating

The susceptibility of land to erosion can be seen as a composite of how easily the substrate can be eroded, and any mitigating effects of the surface vegetation. The higher the risk of erosion the more vulnerable the soil profile and higher the risk of sediment transport to watercourses. By identifying the risk, areas vulnerable to land use change can be targeted for mitigation work or runoff control measures.			
Significant effects	Data used	Example attributes	Indicative scoring
Soil and slope characteristics	JHI Inherent risk of erosion by overland flow	Soil texture, runoff and slope characteristics = prone to erosion Soil texture, runoff and slope characteristics = less prone to erosion	high low
Vegetation preventing erosion	Phase 1 habitat layer (1946)	Sparsely vegetated areas Arable land – regularly bare Dense vegetation (e.g. woodland, heaths, bogs)	high low
Pollination resource			NEA service type: Regulating Supporting
A biotic pollinator is any living organism that moves pollen from the male anthers of a flower to the female stigma of a flower enabling fertilisation. The pollination resource can be seen as the amount of pollen present in an area. Areas poor in pollen producing species are unable to produce enough pollen to support pollinator species. Pollinators are essential for the maintenance of many habitat types and production of insect pollinated crops. Pollination as a service is not often mapped due the relatively small scale of the process. Most common known proxy methods to map pollination involve the use of land cover and land use, pollinator habitat and crop yields.			
Significant effects	Data used	Example attributes	Indicative scoring
Species which affect pollination	Species Borders notable species	Bee species Butterflies & moths Dragonflies (associated with pollinator predation around water)	high medium negative
Species which produce pollen	Species Borders notable species	Flowering plants	high
Indicative pollen presence	Phase 1 habitat layer (1946)	Habitat often contains a high proportion of pollen rich species (e.g. heath, scrub) Habitat often contains some pollen rich species (e.g. Semi-natural grassland) Habitat contains few pollen	medium/high medium low Medium Very low

		rich species (e.g. woodland, improved grassland) *Insect pollinated flowering crop (e.g. Oil seed rape, legumes, potatoes)* *Non-insect pollinated crop (e.g. Silage, Oats, Wheat)*	
Water quantity			NEA service type Regulating
Water quantity regulation is a key ecosystem service as excess water in a natural system can cause flooding events. The regulation of water is complex and is affected by factors such as climate (rainfall), but also less obvious ones such as topography, soil, vegetation and land cover type (such as concrete and tarmac). Soil temporarily stores rain water as it percolates through the system towards rivers and streams, or into the groundwater resource. The ability of soil to perform this function depends on its texture, depth and organic matter content, as well as the overall context of the soil in the landscape. Habitat type, through its link to vegetation type and soil type, has an important influence on water quantity. This is greatly influenced by the structure of the vegetation present and its effect on infiltration. Steep slopes shed water more rapidly than shallow slopes. Steep slopes are also more likely to be in the upper reaches of catchments and are characterised by small streams with rocky banks, which in times of heavy rainfall can quickly rise.			
Significant effects	Data used	Example attributes	Indicative scoring
Vegetation effect on interception	Phase 1 habitat layer (1946)	Dense vegetation (e.g., woodland) Variable density vegetation (e.g., heath, bog) Low density vegetation and vegetation often removed (e.g., arable)	high moderate low
Infiltration and drainage characteristics of the ground	Soil / geology National Soil Inventory Scotland 1:250,000 with HOST classification BGS Superficial 1:50,000 BGS Bedrock 1:50,000	free drainage poor drainage Permeable substrate Impermeable substrate	high low high low
Drainage	Drainage and topography DTM	Gentle slopes Steep slopes	high low
Biodiversity and nature conservation			NEA service type: Regulating and maintenance Provisioning Supporting

<table>
<tr><th colspan="3"></th><th>Cultural</th></tr>
<tr><td colspan="4">Biodiversity is an important supporting ecosystem service that underpins a majority of ecosystem services. Biodiversity describes the range and diversity of species existing and includes genetic diversity within species and between different taxa in any area.
Climax communities of semi-natural habitats that have been present for a long period of time tend to have the highest biodiversity, as over time they can develop specialized niches. The structure of the vegetation both above and below ground has a profound effect on biodiversity. The more complex the structures and the more varied the niches or locations for biodiversity development the greater the diversity of species found in an ecosystem.
The value of a parcel of land for biodiversity and nature conservation can be assessed by considering:
Naturalness – those habitats which have received little modification by humans.
Diversity – The higher the plant community species richness, the higher the diversity within the habitat. This is difficult to accurately compare as some plant communities are intrinsically more species rich than others. Detailed habitat classifications such as Annex I or NVC, which take into account the presence of species and communities, can be added to the broader habitat classifications to model species diversity.
Connectivity – Habitats which are well connected are more likely to support a greater number of organisms that inhabit that particular ecological niche. Fragmented patches (depending on size) can only support smaller populations.
All vegetation types have been scored in this biodiversity layer and then any management and connectivity have been added as modifiers to infer more likelihood of good quality habitat.</td></tr>
<tr><td>Significant effects</td><td>Data used</td><td>Example attributes</td><td>Indicative scoring</td></tr>
<tr><td>Naturalness</td><td>Habitats
Phase 1 habitat layer (1946)</td><td>Semi-natural habitats (e.g. heath, bog, woodland)
Other habitat (e.g. scrub, parkland, bracken)
Intensively managed land (e.g. improved grassland, arable, urban)</td><td>high
medium
low</td></tr>
<tr><td>Diversity</td><td>Species
Borders Notable Species</td><td>Internationally important
Nationally important
Locally important</td><td>high
medium
low</td></tr>
<tr><td></td><td>Habitats
Phase 1 habitat layer (1946)</td><td>Other habitat (e.g. scrub, parkland, bracken)
Intensively managed land (e.g. improved grassland, arable, urban)</td><td>medium
low</td></tr>
<tr><td>Location within the landscape</td><td>Phase 1 habitat layer (1946)</td><td>Well connected habitat
Poorly connected habitat</td><td>high
low</td></tr>
</table>

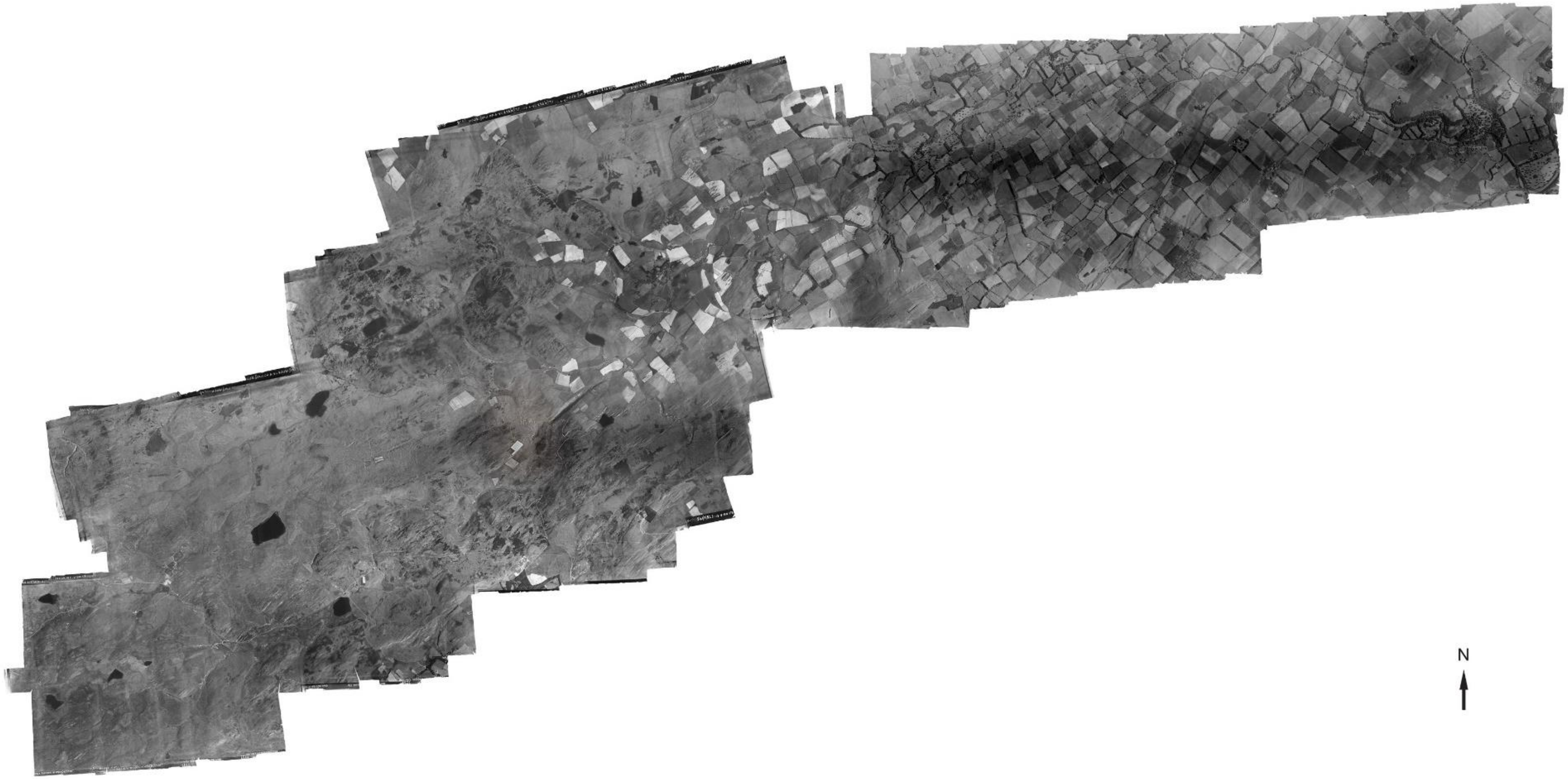
Appendix 7: Ecosystem service provision scores based on the look up tables

Habitat	Water quantity	Water quality	Vegetation Carbon	Soil Carbon	Pollination	Biodiversity	Land erosion risk
Acid grassland - semi-improved	100	150	50	100	150	200	200
Acid grassland - unimproved	100	150	50	150	150	250	100
Blanket bog	200	200	200	250	100	250	50
Bracken - scattered	100	150	150	50	50	150	150
Bracken - continuous	100	150	200	100	50	100	100
Broadleaved parkland/scattered trees	250	150	250	100	100	200	100
Broadleaved woodland - plantation	250	150	250	200	100	200	50
Broadleaved - recently planted	200	100	150	50	0	50	200
Broadleaved woodland - semi-natural	250	250	250	200	100	250	100
Built land	0	0	0	0	0	0	0
Coniferous woodland - plantation	250	150	250	100	0	100	50
Coniferous woodland - recently planted	200	100	150	50	0	50	200
Improved grassland - amenity	50	50	50	150	50	50	150

Habitat	Water quantity	Water quality	Vegetation Carbon	Soil Carbon	Pollination	Biodiversity	Land erosion risk
Cultivated/disturbed land - arable	0	0	0	50	250	200	250
Dry dwarf shrub heath	150	150	200	200	250	200	150
Dry heath/acid grassland	150	150	150	250	100	150	100
Dry modified bog	50	0	100	50	0	0	150
Fen - valley mire	200	150	150	50	0	0	0
Flush and spring - acid/neutral flush	200	150	50	250	100	200	150
Gardens	50	50	50	200	100	200	150
Hedgerow	150	200	200	50	100	100	50
Improved grassland	50	0	50	150	150	50	200
Marsh/marshy grassland	200	150	100	50	50	100	50
Mixed woodland - plantation	250	150	250	150	100	250	50
Neutral grassland - semi-improved	100	150	50	50	50	50	100
Neutral grassland - unimproved	100	150	50	200	50	150	100
Other tall herb and fern	100	50	50	100	0	50	150
Poor semi-improved grassland	100	150	50	100	50	150	100
Quarry	0	0	0	0	0	0	250
Refuse tip	0	0	0	0	50	50	200
Running water	0	0	0	0	50	50	50

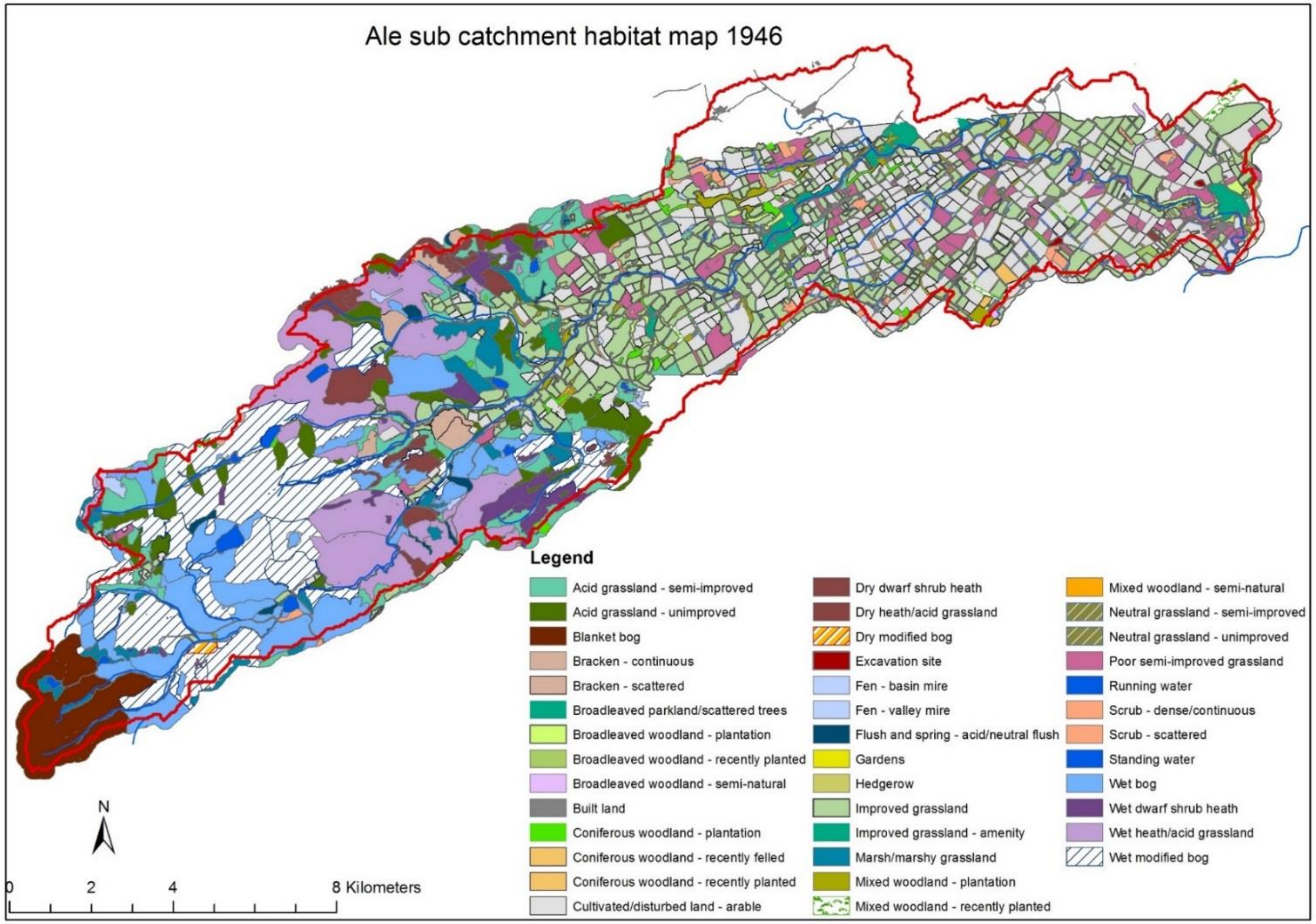
Habitat	Water quantity	Water quality	Vegetation Carbon	Soil Carbon	Pollination	Biodiversity	Land erosion risk
Scrub - dense/continuous	200	200	150	50	50	150	100
Scrub - scattered	150	200	100	150	250	100	150
Standing water	0	0	0	0	50	150	150
Wet bog	200	200	150	250	150	200	50
Wet dwarf shrub heath	150	200	150	200	200	250	100
Wet heath/acid grassland	150	200	150	150	150	250	100
Wet modified bog	100	100	150	200	100	250	200

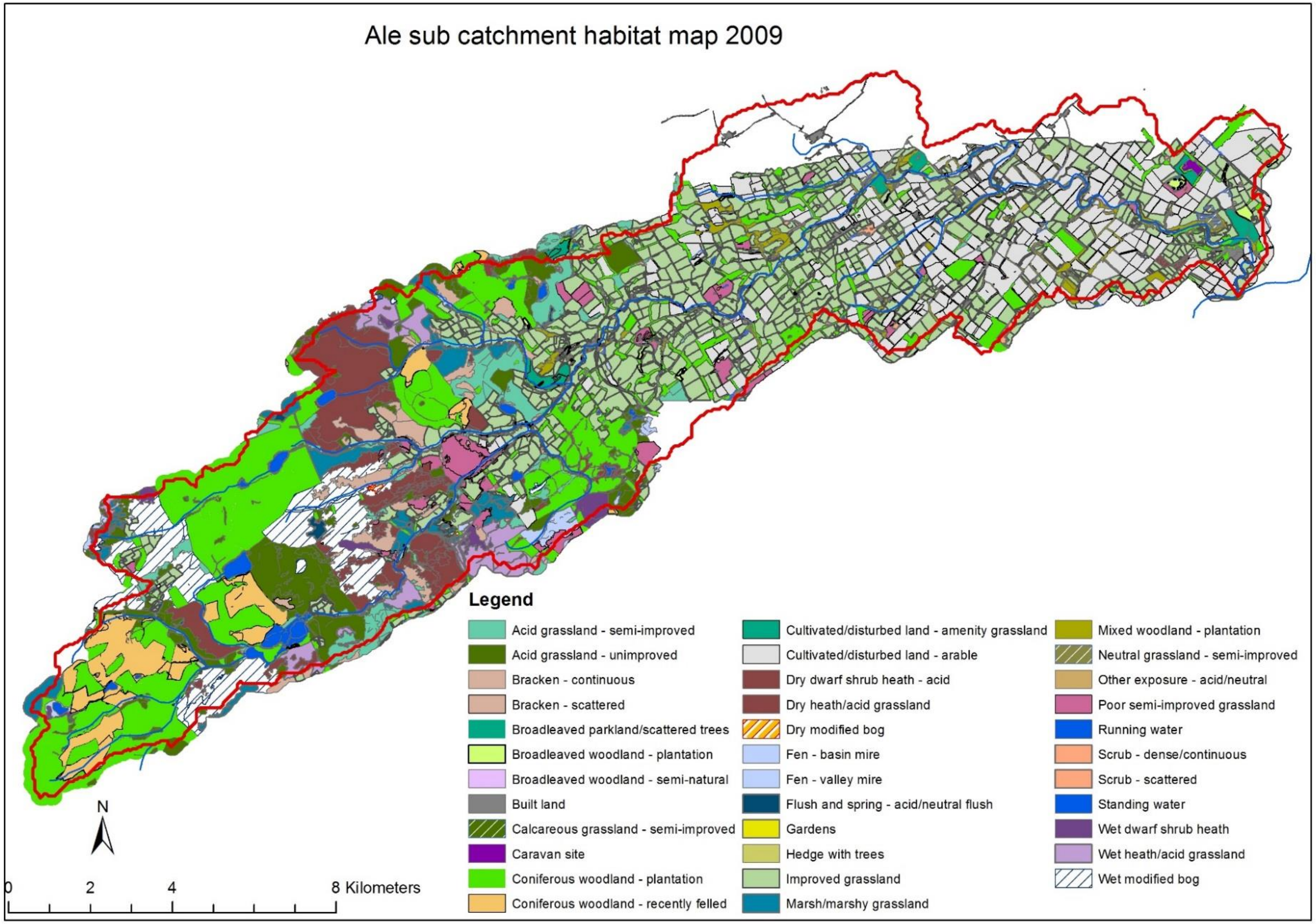
Appendix 8: The Ale catchment 1946 photo mosaic



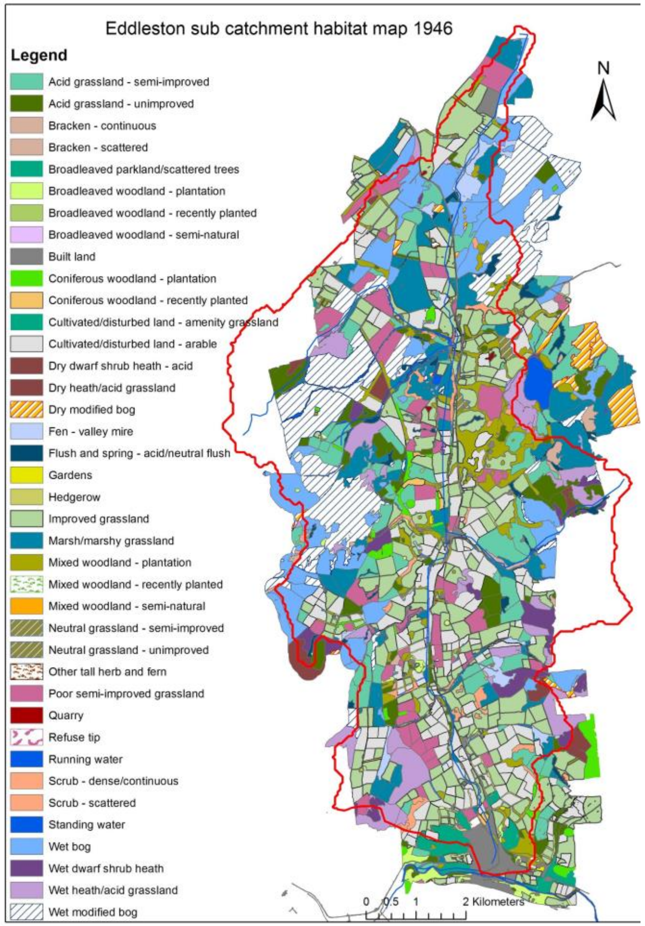
Appendix 9: The Eddleston catchment 1946 photo mosaic

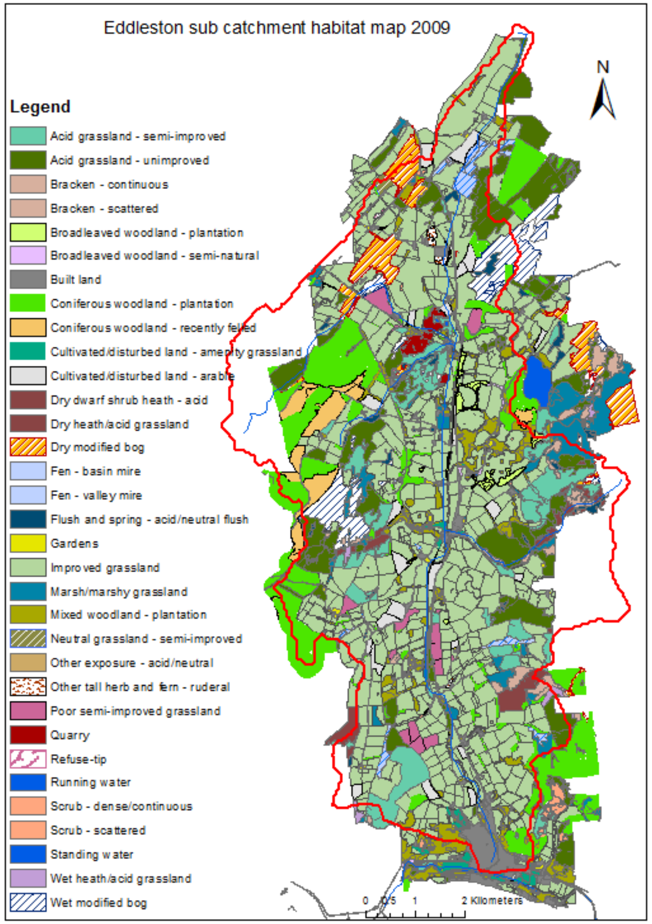
Appendix 10a: 1946 and 2009 habitat maps for the Ale catchment




Ale sub catchment habitat map 2009
Legend
Acid grassland - semi-improved
Acid grassland - unimproved
Bracken - continuous
Bracken - scattered
Broadleaved parkland/scattered trees
Broadleaved woodland - plantation
Broadleaved woodland - semi-natural
Built land
Calcareous grassland - semi-improved
Caravan site
Coniferous woodland - plantation
Coniferous woodland - recently felled
Cultivated/disturbed land - amenity grassland
Cultivated/disturbed land - arable
Dry dwarf shrub heath - acid
Dry heath/acid grassland
Dry modified bog
Fen - basin mire
Fen - valley mire
Flush and spring - acid/neutral flush
Gardens
Hedge with trees
Improved grassland
Marsh/marshy grassland
Mixed woodland - plantation
Neutral grassland - semi-improved
Other exposure - acid/neutral
Poor semi-improved grassland
Running water
Scrub - dense/continuous
Scrub - scattered
Standing water
Wet dwarf shrub heath
Wet heath/acid grassland
Wet modified bog
N
0 2 4 8 Kilometers

Appendix 10b: 1946 and 2009 habitat maps for the Eddleston catchment





Appendix 11a: Error matrix for the Ale catchment

		Reference data (Field verification 2014)																					
		Acid grassland - semi-improved	Acid grassland - unimproved	Bracken - scattered	Bracken - continuous	Broadleaved parkland/ scattered trees	Broadleaved woodland - plantation	Built land	Coniferous woodland - plantation	Cultivated /disturbed land - arable	Dry heath/acid grassland	Hedgerow	Improved grassland	Marsh/marshy grassland	Mixed woodland - plantation	Poor semi-improved grassland	Running water	scrub - dense/continuous	Scrub - scattered	Standing water	Wet bog	Wet heath/acid grassland	Wet modified bog	**Row Total**
2009 habitat map	Acid grassland - semi-improved	5									1													6
	Acid grassland - unimproved	1	1																					2
	Bracken - scattered			3																				3
	Bracken - continuous				3														1					4
	Broadleaved parkland/scattered trees					1																		1
	Broadleaved woodland - plantation						5																	5
	Built land							10																10
	Coniferous woodland - plantation								17															17
	Cultivated/disturbed land - arable									11														11
	Dry heath/acid grassland										1													1
	Hedgerow											2												2
	Improved grassland												14											14
	Marsh/marshy grassland		1						1					3										5
	Mixed woodland - plantation														5									5
	Poor semi-improved grassland	1											1			1								3
	Running water																8							8
	scrub - dense/continuous																	0						0
	Scrub - scattered																		2					2
	Standing water																			2				2
	Wet bog																				2			2
	Wet heath/acid grassland				2								1					1				9		13
	Wet modified bog																						0	0
	Column total	**7**	**2**	**3**	**5**	**1**	**5**	**10**	**18**	**11**	**2**	**2**	**16**	**3**	**5**	**1**	**8**	**1**	**3**	**2**	**2**	**9**	**0**	**116**
	Overall accuracy = 105/116 = 90.5%																							

Appendix 11b: Error matrix for the Eddleston catchment

2009 habitat map	Reference data (Land Cover of Scotland 1988 air photo interpretation) → Acid grassland - semi-improved	Acid grassland - unimproved	Broadleaved parkland/ scattered trees	Broadleaved woodland - plantation	Built land	Coniferous woodland - plantation	Cultivated /disturbed land - arable	Dry heath/acid grassland	Improved grassland	Marsh/marshy grassland	Mixed woodland - plantation	Poor semi-improved grassland	Running water	Scrub - scattered/ dense/continuous	Standing water	Wet bog	Wet heath/acid grassland	Wet modified bog	Row Total
Acid grassland - semi-improved	8								1										9
Acid grassland - unimproved		2						1											3
Broadleaved parkland/scattered trees			1																1
Broadleaved woodland - plantation				3															3
Built land					5														5
Coniferous woodland - plantation						11													11
Cultivated/disturbed land - arable							2												2
Dry heath/acid grassland								2											2
Improved grassland									14										14
Marsh/marshy grassland	1									1									2
Mixed woodland - plantation						1					7								8
Poor semi-improved grassland							1		1			3							5
Running water													4						4
Scrub - scattered/dense/continuous														2					2
Standing water															1				1
Wet bog								1								2			3
Wet heath/acid grassland								1									3		4
Wet modified bog	1																	3	4
Column total	10	2	1	3	5	12	3	5	16	1	7	3	4	2	1	2	3	3	83

Overall accuracy: = 74/83 = 88.1%

Appendix 12 a: Overview of habitat changes in the Ale catchment

Habitat types recorded in 1946 only:	Habitat types recorded in 2009 only:	Habitat types recorded in both years, change in area less than 0.5% of the total catchment area	Habitat types recorded in both years, declining areas of which change is more than 0.5% of the total catchment area	Habitat types recorded in both years, increasing areas in which change is more than 0.5% of the total catchment area
-Blanket bog -Wet bog -Excavation sites -Broadleaved woodland – recently planted -Coniferous woodland – recently planted -Mixed woodland – recently planted -Mixed woodland – semi natural -Neutral grassland – unimproved	-Caravan site -Other exposures – acid/neutral -Calcareous grassland – semi-improved	<u>**Slightly decreased**</u> -Broadleaved woodland – semi-natural -Dry modified bog -Flush and spring – acid/neutral flush -Scrub – dense/continuous -Scrub – scattered <u>**Slightly Increased**</u> -Built land -Gardens -Fen-basin mire -Fen – valley mire -Improved grassland Amenity -Marshy grassland -Mixed woodland - plantation -Running water i.e. rivers and streams	-Acid grassland – semi-improved -Bracken – scattered -Broadleaved woodland – parkland/scattered trees -Broadleaved woodland plantation -Neutral grassland – semi-improved -Poor semi-improved grassland -Wet dwarf shrub heath -Wet heath/acid grassland -Wet modified bog	-Acid grassland – unimproved -Bracken – continuous -Coniferous woodland plantation -Coniferous woodland - recently felled -Dry dwarf shrub heath -Dry heath/acid grassland -Improved grassland -Standing water

Appendix 12 b: Overview of habitat changes in the Eddleston catchment

Habitat types recorded in 1946 only:	Habitat types recorded in 2009 only:	Habitat types recorded in both years, change in area less than 0.5% of the total catchment area	Habitat types recorded in both years, declining areas of which change is more than 0.5% of the total catchment area	Habitat types recorded in both years, increasing areas in which change is more than 0.5% of the total catchment area
-Broadleaved parkland/scattered trees -Broadleaved woodland – recently planted -Coniferous woodland – recently planted -Mixed woodland – recently planted -Neutral grassland – unimproved -Wet dwarf shrub heath	-Coniferous woodland – recently felled	**<u>Slightly decreased</u>** -Broadleaved woodland – plantation -Amenity grassland -Dry heath/acid grassland -Fen – valley mire -Flush and spring –acid/neutral flush -Neutral grassland – semi-improved -Refuse tip -Scrub – dense/continuous **<u>Slightly increased</u>** -Bracken – continuous -Broadleaved woodland – semi-natural -Built land -Dry dwarf shrub heath -Gardens -Standing water -Quarry sites	-Acid grassland – semi-improved -Cultivated/disturbed land – arable -Marshy grassland -Poor semi-improved grassland -Scrub – scattered -Wet heath/acid grassland -Wet modified bogs	-Acid grassland – unimproved -Bracken – scattered -Coniferous woodland plantations -Dry modified bogs -Improved grassland -Mixed woodland plantations

Appendix 13a: Areal extent of habitats in the Ale catchment between 1946 and 2009

		Year 1946		Year 2009		Change	% change of	Direction of	Level of statistical
	Habitat type	**Area (ha)**	**%**	**Area (Ha)**	**%**	**(ha)**	**total catchment**	**change**	**significance of change**
	Acid grassland - semi-improved	924.81	5.42	639.94	3.75	-284.87	-1.67	Decrease	***
	Acid grassland - unimproved	712.77	4.18	1047.22	6.14	334.45	1.96	Increased	***
	Blanket bog	630.09	3.69	0.00	0.00	-630.09	-3.69	Decreased	n/a
	Bracken - continuous	77.64	0.46	352.31	2.06	274.67	1.61	Increased	*
	Bracken - scattered	125.45	0.74	29.74	0.17	-95.71	-0.56	Decreased	*
	Broadleaved parkland/scattered trees	206.69	1.21	57.92	0.34	-148.77	-0.87	Decreased	**
	Broadleaved woodland - plantation	200.85	1.18	46.68	0.27	-154.17	-0.90	Decreased	***
	Broadleaved woodland - recently planted	8.51	0.05	0.00	0.00	-8.51	-0.05	Decreased	n/a
	Broadleaved woodland - semi-natural	15.55	0.09	9.54	0.06	-6.01	-0.04	Decreased	**
	Built land	283.99	1.66	347.02	2.03	63.03	0.37	Increased	—
	Calcareous grassland - semi-improved	0.00	0.00	3.28	0.02	3.28	0.02	Increased	n/a
	Caravan site	0.00	0.00	11.61	0.07	11.61	0.07	Increased	n/a
	Coniferous woodland - plantation	121.24	0.71	2970.23	17.41	2848.99	16.70	Increased	***
	Coniferous woodland - recently felled	24.86	0.15	632.85	3.71	607.98	3.56	Increased	**
	Coniferous woodland - recently planted	0.99	0.01	0.00	0.00	-0.99	-0.01	Decreased	n/a
	Cultivated/disturbed land - arable	2857.99	16.75	2497.12	14.63	-360.87	-2.11	Decreased	***
	Dry dwarf shrub heath	139.49	0.82	399.89	2.34	260.40	1.53	Increased	**
	Dry heath/acid grassland	309.53	1.81	730.57	4.28	421.04	2.47	Increased	*
	Dry modified bog	15.98	0.09	4.51	0.03	-11.48	-0.07	Decreased	—
	Excavation site	12.85	0.08	0.00	0.00	-12.85	-0.08	Decreased	*
	Fen - basin mire	27.44	0.16	34.79	0.20	7.36	0.04	Increased	*
	Fen - valley mire	48.78	0.29	81.58	0.48	32.80	0.19	Increased	—
	Flush and spring - acid/neutral flush	79.81	0.47	35.86	0.21	-43.95	-0.26	Decreased	**
	Gardens	1.63	0.01	10.95	0.06	9.32	0.05	Increased	*
	Improved grassland	3021.55	17.71	4219.26	24.73	1197.71	7.02	Increased	***
	Improved grassland - amenity	10.12	0.06	73.27	0.43	63.15	0.37	Increased	—
	Marsh/marshy grassland	386.56	2.27	415.03	2.43	28.47	0.17	Increased	***
	Mixed woodland - plantation	268.12	1.57	321.24	1.88	53.12	0.31	Increased	***
	Mixed woodland - recently planted	48.99	0.29	0.00	0.00	-48.99	-0.29	Decreased	*
	Mixed woodland - semi-natural	5.60	0.03	0.00	0.00	-5.60	-0.03	Decreased	n/a
	Neutral grassland - semi-improved	135.21	0.79	52.06	0.31	-83.14	-0.49	Decreased	**
	Neutral grassland - unimproved	15.94	0.09	0.00	0.00	-15.94	-0.09	Decreased	*
	Other exposure - acid/neutral	0.00	0.00	0.23	0.00	0.23	0.00	Increased	n/a
	Poor semi-improved grassland	730.42	4.28	386.68	2.27	-343.74	-2.01	Decreased	***
	Scrub - dense/continuous	85.74	0.50	43.43	0.25	-42.31	-0.25	Decreased	**
	Scrub - scattered	81.84	0.48	17.27	0.10	-64.57	-0.38	Decreased	***
	Standing water	117.17	0.69	198.86	1.17	81.69	0.48	Increased	**
	Wet bog	1387.77	8.13	0.00	0.00	-1387.77	-8.13	Decreased	***
	Wet dwarf shrub heath	322.44	1.89	108.93	0.64	-213.51	-1.25	Decreased	—
	Wet heath/acid grassland	1489.42	8.73	365.41	2.14	-1124.01	-6.59	Decreased	**
	Wet modified bog	2129.52	12.48	918.09	5.38	-1211.43	-7.10	Decreased	—
	Total catchment area	**17063.4**	**100**	**17063.4**	**100**	**0**	**0**		
NB:									
n/a	not possible to test statistically as number of counts is too small								
—	Statistically Insignificant								
*	p<0.05								
**	p<0.01								
***	p<0.001								

Appendix 13b: Areal extent of habitats in the Eddleston catchment between 1946 and 2009

Habitat type	Year 1946 Area (ha)	Year 1946 %	Year 2009 Area (ha)	Year 2009 %	Change (ha)	total catchment % change	Direction of change	Level of Statistical significance
Acid grassland - semi-improved	609.39	7.69	429.78	5.42	-179.61	-2.27	Decrease	—
Acid grassland - unimproved	330.36	4.17	819.52	10.34	489.16	6.17	Increase	***
Bracken - continuous	43.07	0.54	63.92	0.81	20.84	0.26	Increase	—
Bracken - scattered	2.22	0.03	43.12	0.54	40.91	0.52	Increase	***
Broadleaved parkland/scattered trees	61.32	0.77	0.00	0.00	-61.32	-0.77	Decrease	**
Broadleaved woodland - plantation	67.69	0.85	44.14	0.56	-23.54	-0.30	Decrease	**
Broadleaved woodland - recently planted	0.56	0.01	0.00	0.00	-0.56	-0.01	Decrease	n/a
Broadleaved woodland - semi-natural	1.44	0.02	8.38	0.11	6.94	0.09	Increase	***
Built land	241.61	3.05	283.30	3.57	41.70	0.53	Increase	—
Coniferous woodland - plantation	92.67	1.17	868.28	10.95	775.61	9.78	Increase	**
Coniferous woodland - recently planted	14.00	0.18	0.00	0.00	-14.00	-0.18	Decrease	*
Coniferous woodland - recently felled	0.00	0.00	150.60	1.90	150.60	1.90	Increase	**
Cultivated/disturbed land - amenity grassland	44.34	0.56	40.23	0.51	-4.11	-0.05	Decrease	**
Cultivated/disturbed land - arable	819.22	10.34	152.07	1.92	-667.16	-8.42	Decrease	*
Dry dwarf shrub heath	9.20	0.12	45.81	0.58	36.62	0.46	Increase	—
Dry heath/acid grassland	86.90	1.10	65.51	0.83	-21.38	-0.27	Decrease	—
Dry modified bog	114.29	1.44	219.46	2.77	105.17	1.33	Increase	—
Fen- valley mire	92.87	1.17	77.82	0.98	-15.05	-0.19	Decrease	—
Flush and spring - acid neutral flush	96.23	1.21	70.07	0.88	-26.16	-0.33	Decrease	—
Gardens	3.99	0.05	20.92	0.26	16.94	0.21	Increase	***
Improved grassland	1497.51	18.89	3333.32	42.05	1835.81	23.16	Increase	***
Marsh/marshy grassland	651.48	8.22	256.95	3.24	-394.53	-4.98	Decrease	***
Mixed woodland - plantation	351.01	4.43	398.41	5.03	47.40	0.60	Increase	***
Mixed woodland - recently planted	0.65	0.01	0.00	0.00	-0.65	-0.01	Decrease	n/a
Mixed woodland - semi-natural	0.27	0.00	0.00	0.00	-0.27	0.00	Decrease	n/a
Neutral grassland - semi-improved	19.96	0.25	11.37	0.14	-8.59	-0.11	Decrease	—
Neutral grassland - unimproved	15.83	0.20	0.00	0.00	-15.83	-0.20	Decrease	—
Other tall herb and fern	2.71	0.03	13.85	0.17	11.14	0.14	Increase	—
Poor semi-improved grassland	431.08	5.44	96.47	1.22	-334.61	-4.22	Decrease	*
Quarry	3.48	0.04	36.50	0.46	33.02	0.42	Increase	—
Refuse tip	3.33	0.04	1.60	0.02	-1.73	-0.02	Decrease	n/a
Scrub - dense/continuous	23.59	0.30	21.20	0.27	-2.38	-0.03	Decrease	***
Scrub - scattered	63.04	0.80	9.98	0.13	-53.06	-0.67	Decrease	**
Standing water	43.08	0.54	50.47	0.64	7.39	0.09	Increase	—
Wet bog	745.81	9.41	0.00	0.00	-745.81	-9.41	Decrease	***
Wet dwarf shrub heath	148.16	1.87	0.00	0.00	-148.16	-1.87	Decrease	**
Wet heath/acid grassland	374.44	4.72	20.90	0.26	-353.54	-4.46	Decrease	—
Wet modified bog	819.77	10.34	272.63	3.44	-547.15	-6.90	Decrease	—
Total	**7926.6**	**100**	**7926.6**	**100**	**0**	**0**		

NB:

n/a not possible to test statistically as sample size is too small

— Statistically Insignificant

* p<0.05

** p<0.01

*** p<0.001

Appendix 14: Landscape metrics computed in FRAGSTAT

Measure	Description	Unit/comment
Number of Patches (NUMP)	Counts a) the total number of patches (landscape level) or b) the number of patches of the focal class (class level).	Unit: None Range: NUMP > 0
Mean Patch Size (MPS)	The average of the individual area coverage of a) all patches in the landscape (landscape level) or b) all patches of the focal class (class level). Small values can be indicative of fragmentation and little core area.	Unit: ha Range: MPS > 0
Patch size standard deviation	Habitat type distribution statistics summaries Can be measured at both landscape and habitat type level	PSSD > 0
Patch size coefficient of variance	Habitat type distribution statistics summaries	PSCV > 0
Mean Proximity Index (MPI)	The mean proximity index measures the degree of isolation and fragmentation of the corresponding patch type. It takes into consideration the number and size of patches of the same type within a certain radius around the focal patch. This allows to distinguish between sparse distributions of small patches or a dense cluster of large patches. Thus measuring the degree of patch isolation and degree of fragmentation. It refers to the tendency for patches to be relatively isolated in space i.e. distant from other patches of the same or similar class.	Unit: None Range: MPI ≥ 0 Is “0” if for all focal patches no patch of the same type is within the specified radius.
Connectance Index	Connectance is defined on the number of functional joinings between patches of the corresponding patch type, where each pair of patches is either connected or not based on a user-specified distance criterion. Joinings among all	Unit: percent Range: 0 ≤ Connect ≤ 100 Is 0 when none of the patches of the focal class are connected within the specified threshold distance of another patch of the same type. Is = 100 when every

Measure	Description	Unit/comment
	patches of the same type. Connectance is reported as a percentage of the maximum possible connectance given the number of patches.	patch of the focal class is connected
Edge Density (ED)	Measures the amount of patch border present per hectare of patch. Large numbers are indicative of a) irregularly-shaped patches and/or b) small, disjunct patches. At landscape level: Considers edges between all patch types. At class level: Considers edges between the focal patch and all other patch types.	Unit: m/ha Range: ED ≥ 0 Is "0" if the entire landscape consists of one patch and landscape borders have been defined as "non-edges".
Mean shape Index	Shape complexity relates to the geometry of patches-- whether they tend to be simple and compact, compared to a standard shape (square) of the same size,	Units: None Range: SHAPE ≥1 Is = 1 when patch is square and increases as patch shape becomes more irregular higher values mean greater shape complexity
Shannon's Diversity Index (SHDI)	Measure of the diversity of patch types in the landscape. The value increases when a) the number of patch types increases and/or when b) when area becomes more evenly distributed among patch types	Unit: None Range: SHDI ≥ 0 Is "0" if there is only one patch

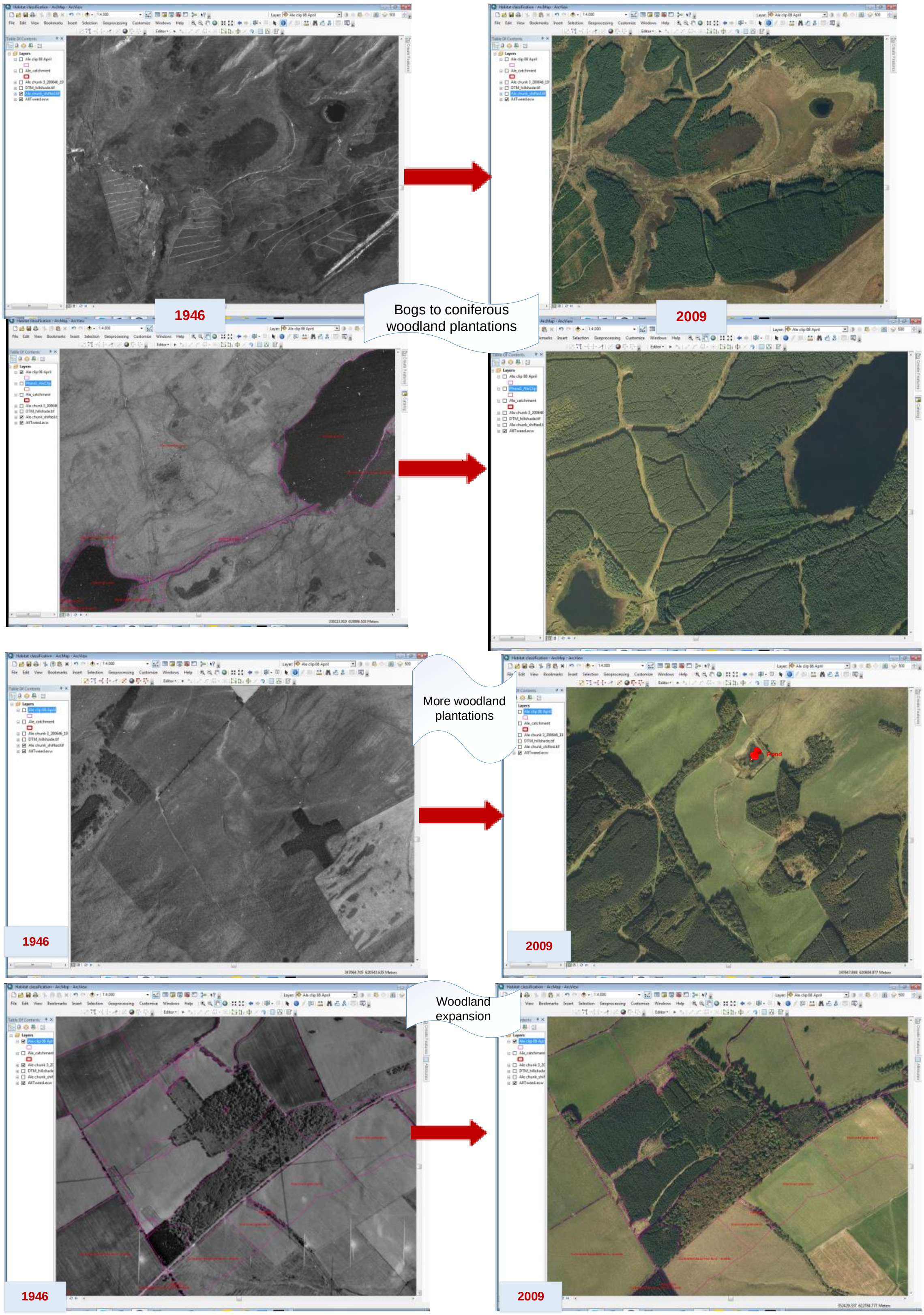
Appendix 15a: Habitat pattern metrics for the Ale catchment:

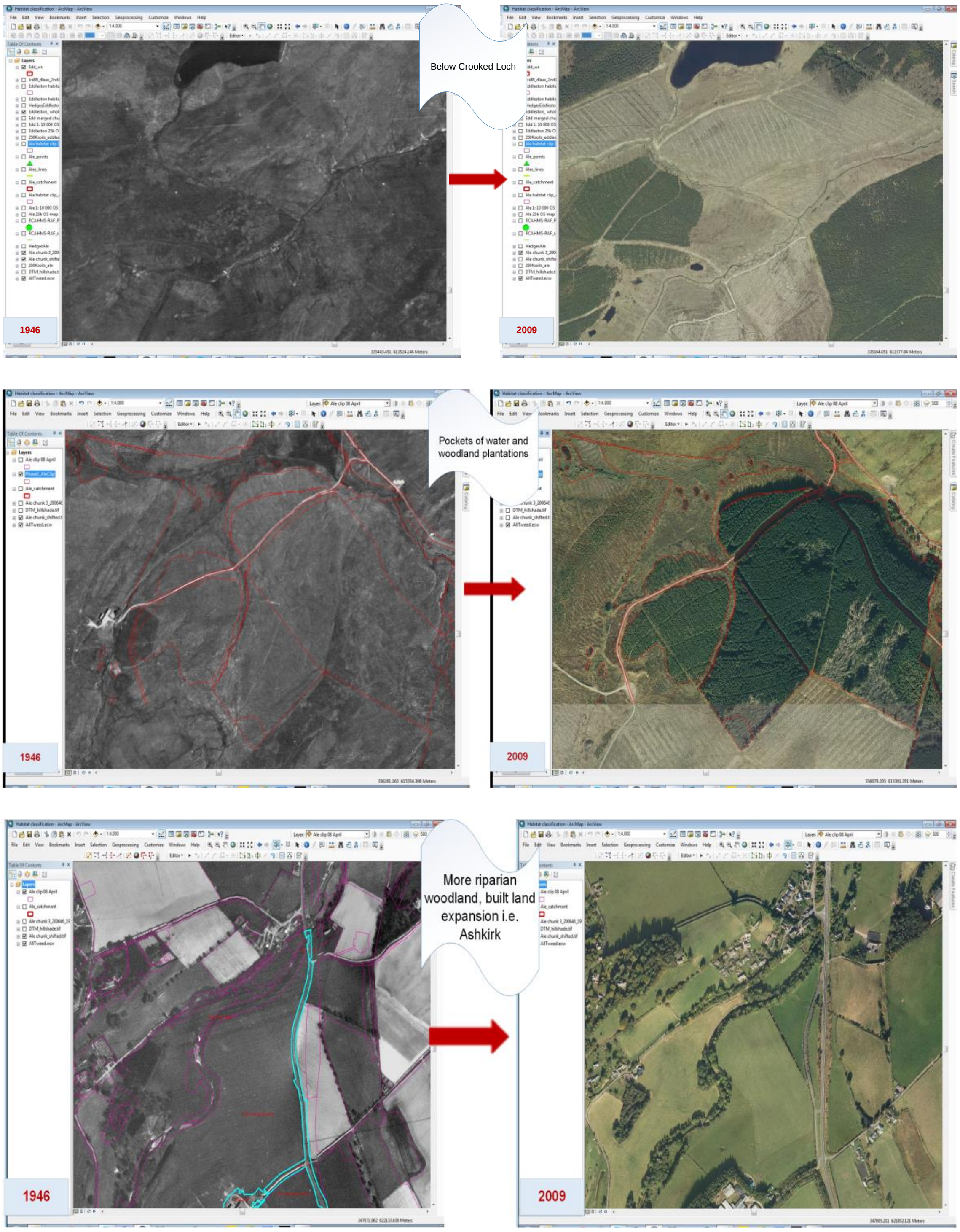
	Number of habitat Patches		Mean patch size (ha)		Patch size standard deviation		Edge Density		Mean Proximity Index		Connectance Index (%)	
Habitat type	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**
Acid grassland - semi-improved	68	202	13.60	3.17	12.77	11.11	9.55	8.22	263.94	126.63	2.93	2.40
Acid grassland - unimproved	53	362	13.44	2.89	13.89	14.74	6.93	11.96	117.71	97.06	3.43	2.00
Bracken - continuous	6	80	12.94	4.40	12.55	9.44	0.61	5.30	0.00	70.79	0.00	3.55
Bracken - scattered	5	8	25.09	3.72	17.43	7.15	0.92	0.30	0.08	0.00	7.14	0.00
Broadleaved parkland/scattered trees	11	24	18.79	2.41	14.58	6.94	1.94	0.58	349.84	0.00	11.54	0.00
Broadleaved woodland - plantation	100	81	2.00	0.58	3.41	1.41	7.22	1.40	95.29	1.70	2.72	4.35
Broadleaved woodland - semi-natural	4	34	3.89	0.28	0.97	0.39	0.35	0.63	8.44	11.32	26.67	27.45
Coniferous woodland - plantation	67	1254	1.81	2.37	2.03	22.21	3.51	27.36	2.70	321.07	2.28	1.53
Cultivated/disturbed land - arable	421	432	6.79	5.78	4.67	6.15	29.70	20.31	483.65	862.06	2.97	3.92
Dry dwarf shrub heath	11	108	12.68	3.70	10.64	8.35	1.79	5.75	77.27	91.80	12.73	5.95
Dry heath/acid grassland	10	109	30.95	6.70	31.89	45.03	6.49	5.12	1219.38	731.53	11.97	5.85
Fen - basin mire	9	36	3.05	0.97	3.29	2.05	0.54	0.95	10.28	3.65	2.22	2.21
Fen - valley mire	7	14	6.97	5.83	7.81	15.64	0.77	0.93	9.19	0.80	7.14	13.89
Flush and spring - acid/neutral flush	13	18	6.13	1.99	4.61	3.60	1.40	0.82	0.03	0.24	1.28	5.71
Improved grassland	511	1516	5.91	2.79	5.39	3.75	34.34	40.82	646.79	2167.69	2.52	2.72
Marsh/marshy grassland	42	94	9.20	4.41	11.58	10.19	4.80	4.95	57.23	51.25	2.73	6.98
Mixed woodland - plantation	112	426	2.39	0.75	3.61	1.63	8.60	10.78	26.13	72.00	2.66	5.15
Neutral grassland - semi-improved	58	53	2.33	0.98	1.92	1.87	4.57	1.33	7.56	4.64	3.42	6.21
Poor semi-improved grassland	129	146	5.66	2.65	5.02	7.37	9.75	4.88	46.96	152.42	2.74	3.77
Scrub - dense/continuous	25	94	3.43	0.46	5.39	1.04	1.45	1.46	13.06	3.39	4.43	4.86
Scrub - scattered	29	61	2.82	0.28	2.36	0.36	1.96	0.99	5.77	2.19	2.15	19.79
Wet dwarf shrub heath	25	13	12.89	8.38	18.15	13.68	3.33	1.17	91.76	2.82	6.32	5.56
Wet heath/acid grassland	20	23	74.47	15.89	101.94	37.05	1.97	2.95	109.03	377.04	21.21	11.11
Wet modified bog	39	39	54.60	23.54	83.32	74.31	9.59	6.24	3252.62	1214.79	9.12	10.99

Appendix 15b: Habitat pattern metrics for the Eddleston catchment

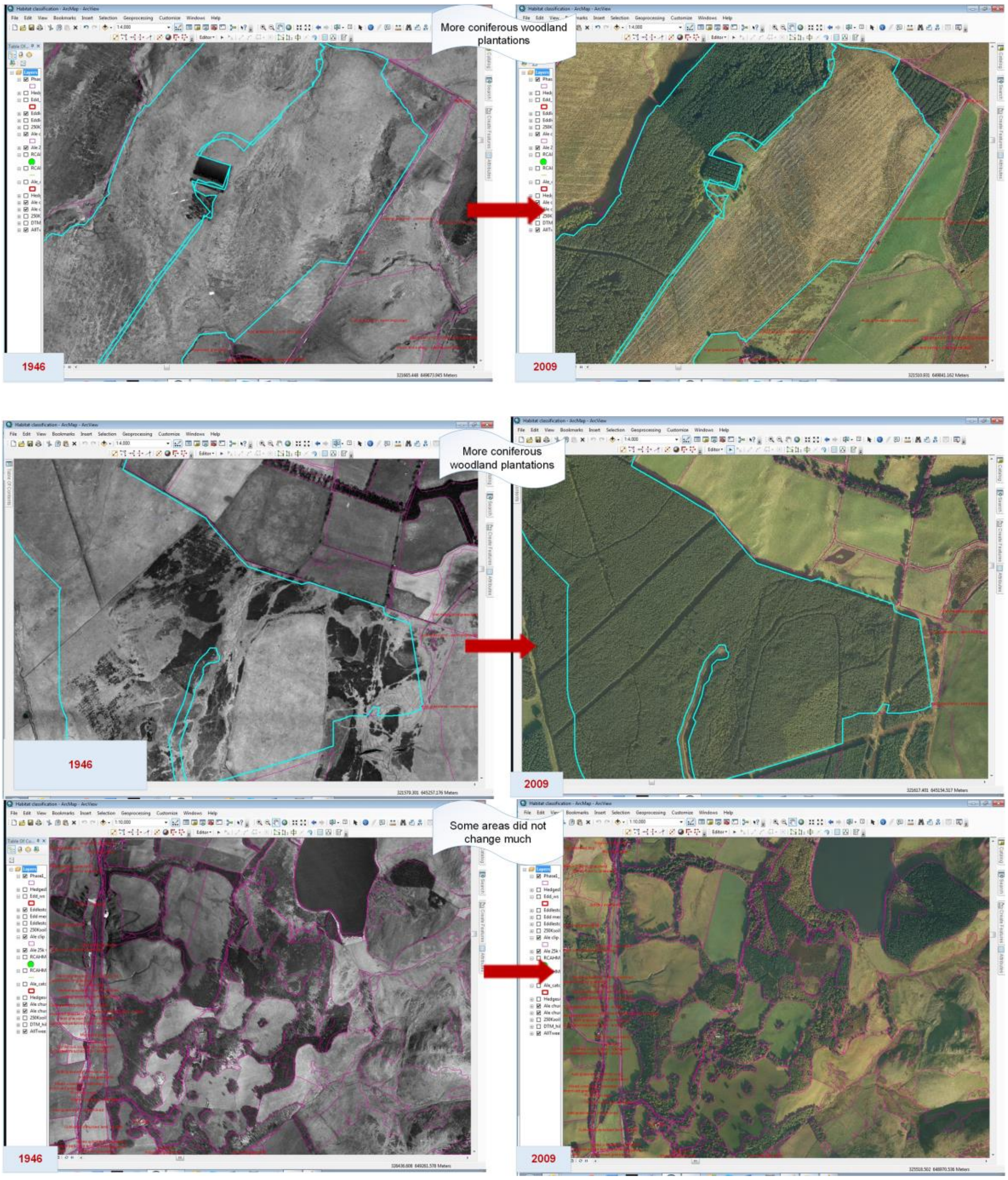
	Number of habitat		Mean patch size (ha)		Patch size standard		Edge Density		Mean Proximity Index		Connectance Index (%)	
Habitat type	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**	**1946**	**2009**
Acid grassland - semi-improved	81	74	7.62	5.89	6.07	12.57	14.95	11.01	68.04	97.20	2.92	4.41
Acid grassland - unimproved	80	189	4.13	4.34	4.95	10.87	11.80	19.34	18.14	127.99	3.06	3.50
Bracken - continuous	12	16	3.59	3.99	3.62	5.33	1.97	2.50	0.88	13.54	3.03	12.09
Broadleaved woodland - plantation	37	47	1.83	0.94	2.87	1.57	4.16	2.21	10.69	24.66	5.24	7.69
Coniferous woodland - plantation	30	323	3.09	2.67	5.54	11.50	3.20	15.59	2.49	177.48	5.03	3.64
Cultivated/disturbed land - arable	192	28	4.27	5.43	2.80	3.89	21.27	4.19	107.40	9.47	5.13	14.04
Dry dwarf shrub heath - acid	2	16	4.59	2.86	5.63	2.43	0.33	1.92	45.85	21.49	100.00	22.86
Dry heath/acid grassland	6	4	14.48	16.38	10.60	18.29	1.53	0.92	0.00	16.66	0.00	16.67
Dry modified bog	15	37	7.62	5.93	11.46	13.44	2.27	4.24	4.41	32.98	9.52	6.67
Fen - valley mire	11	19	8.44	4.10	8.41	5.26	3.62	2.78	11.28	11.01	16.48	18.68
Flush and spring - acid/neutral flush	37	29	2.60	2.42	1.92	2.10	6.37	4.65	5.22	8.99	3.46	3.45
Improved grassland	291	795	5.12	4.22	4.97	6.47	36.17	53.85	211.08	3163.49	4.31	5.99
Marsh/marshy grassland	76	95	8.57	2.70	11.82	7.90	14.95	8.04	105.42	29.78	2.57	3.60
Mixed woodland - plantation	134	508	2.62	0.78	4.21	2.05	18.73	19.49	118.90	71.98	3.61	4.04
Neutral grassland - semi-improved	19	4	1.05	2.84	0.98	4.99	1.97	0.46	2.85	0.96	6.49	20.00
Other tall herb and fern	2	13	1.36	1.07	0.64	1.77	0.20	0.58	5.44	39.92	50.00	50.00
Poor semi-improved grassland	58	17	7.43	5.67	7.80	5.17	10.53	2.39	40.50	19.04	4.15	15.24
Scrub - dense/continuous	14	27	1.68	0.79	1.30	1.45	1.47	1.27	5.07	9.16	14.10	35.06
Scrub-scattered	24	23	2.63	0.43	2.70	0.59	3.59	0.79	2.81	14.10	5.00	6.06
Wet heath/acid grassland	30	7	12.48	2.99	13.76	2.70	7.19	0.86	92.31	14.40	4.56	9.52
Wet modified bog	37	26	44.31	10.46	135.77	31.54	11.10	5.09	494.29	1.70	7.94	13.68

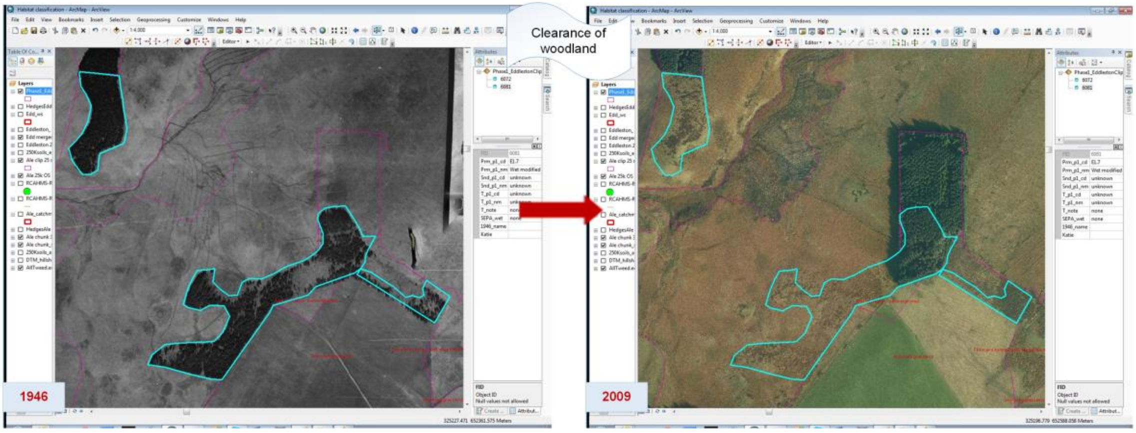
Appendix 16a: Pictorial outline of observed habitat changes in the Ale catchment

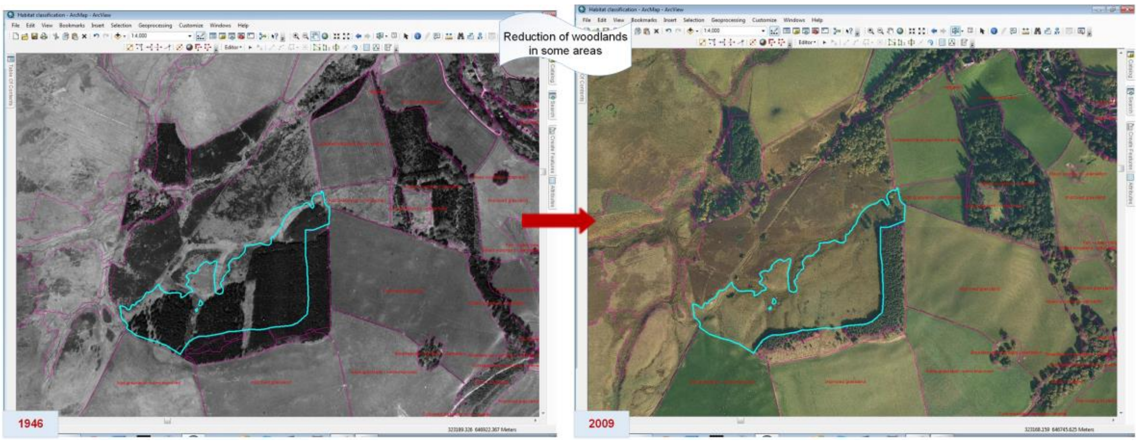


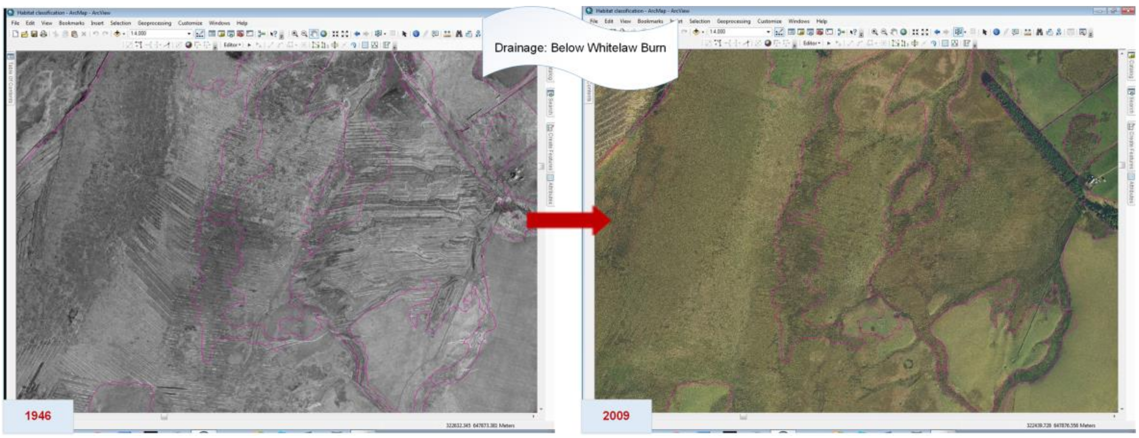

Below Crooked Loch
1946
2009
Pockets of water and woodland plantations
1946
2009
More riparian woodland, built land expansion i.e. Ashkirk
1946
2009

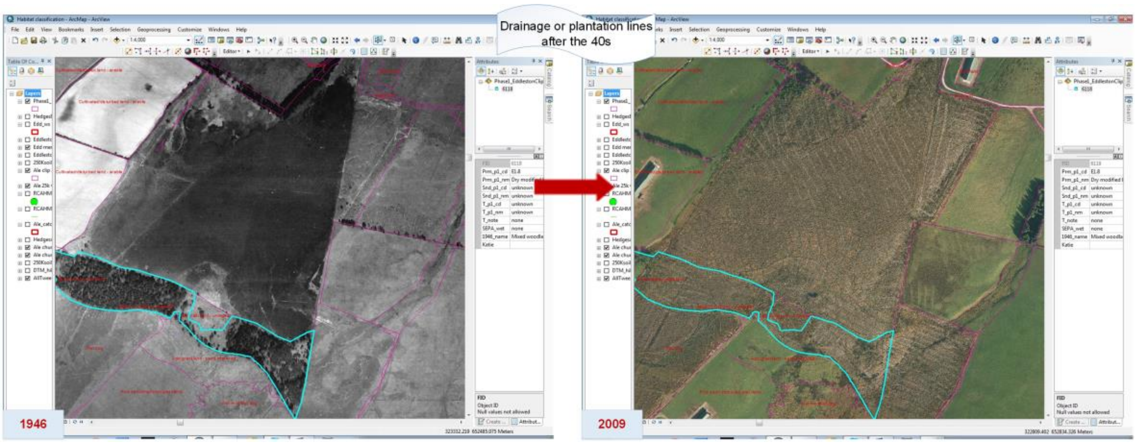
Appendix 16b: Pictorial Outline of observed habitat changes in the Eddleston catchment



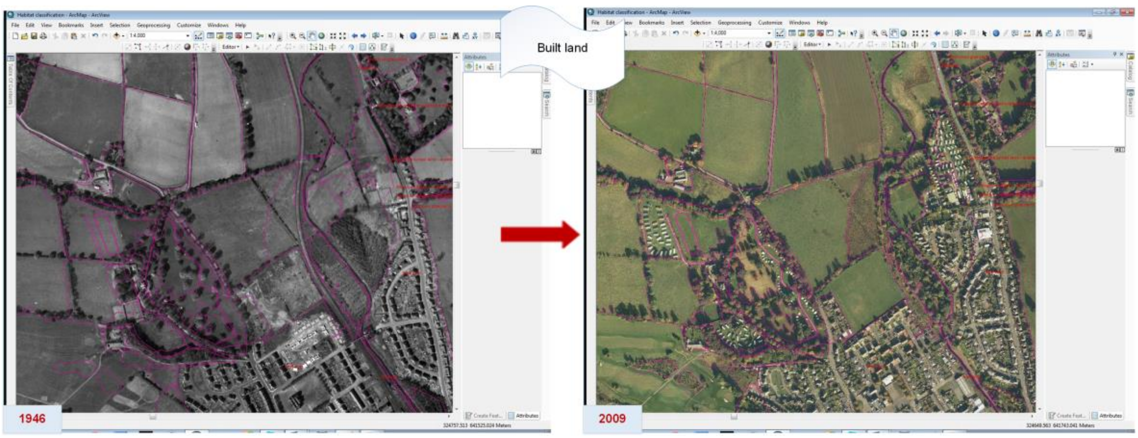

Clearance of
woodland
1946
2009


Reduction of woodlands
in some areas
1946
2009


Drainage: Below Whitelaw Burn
1946
2009


Drainage or plantation lines
after the 40s
1946
2009


Other observed changes i.e. more built land, woodland, small ponds
1946
2009


Built land
1946
2009

Appendix 17a: Ecosystem services maps zonal statistics for the Ale catchment

Ale catchment

SENCE look up table scores		Biodiversity			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	2958.3	3928.39	16.95	22.51
50-150	Medium	3863.83	7085.18	22.14	40.60
150-250	High	4489.59	4380.57	25.73	25.10
Above 250	Very high	6139.5	2057.08	35.18	11.79
	Total	**17451.22**	**17451.22**	**100**	**100**

SENCE look up table scores		Pollination			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	638.99	845.4	3.66	4.84
50-150	Medium	5496.18	8101.77	31.49	46.43
150-250	High	5773.76	3858.54	33.09	22.11
Above 250	Very high	5542.29	4645.52	31.76	26.62
	Total	**17451.2**	**17451.22**	**100**	**100**

SENCE look up table scores		Vegetation carbon			
Relative values	Qualitative scale	Area (Ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	5509.38	3841.79	31.57	22.01
50-150	Medium	5761.94	2425.20	33.02	13.90
150-250	High	3727.11	4677.26	21.36	26.80
Above 250	Very high	2452.80	6506.98	14.06	37.29
	Total	**17451.23**	**17451.23**	**100**	**100**

SENCE look up table scores		Land erosion risk			
		Area (ha)		Percentage (%)	
Relative values	Qualitative scale	1946	2009	1946	2009
Below 50	Low	41.28	24.01	23.68	13.78
50-150	Medium	40.08	31.63	22.99	18.15
150-250	High	57.79	79.93	33.16	45.86
Above 250	Very high	35.15	38.73	20.17	22.22
	Total*	**174.3**	**174.3**	**100**	**100**

SENCE look up table scores		Soil carbon			
		Area (ha)		Percentage (%)	
Relative values	Qualitative scale	1946	2009	1946	2009
Below 50	Low	3745.59	2135.52	21.46	12.24
50-150	Medium	2547.45	8125.68	14.60	46.56
150-250	High	3967.91	4165.60	22.74	23.87
Above 250	Very high	7190.27	3024.42	41.20	17.33
	Total	**17451.22**	**17451.22**	**100**	**100**

SENCE look up table scores		Water quality regulation			
		Area (ha)		Percentage (%)	
Relative values	Qualitative scores	1946	2009	1946	2009
Below 50	Low	205.33	1682.60	1.18	9.64
50-150	Medium	4432.69	6039.79	25.40	34.61
150-250	High	7731.11	5234.98	44.30	30.00
Above 250	Very high	5082.03	4493.79	29.12	25.75
	Total	**17451.16**	**17451.16**	**100**	**100**

*: Small total catchment area due to coarse scale (1:250 000) input soil data used in mapping this ecosystem service

SENCE look up tables scores		Flood control			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	1273.32	361.80	7.30	2.07
50-150	Medium	4763.51	3400.19	27.30	19.48
150-250	High	7082.67	6622.11	40.59	37.95
Above 250	Very high	4331.66	7067.12	24.82	40.50
	Total	**17451.16**	**17451.22**	**100**	**100**

NB: Crops, livestock and timber production ecosystem servcies not included as zonal statistical counts were not done for these.

Appendix 17b: Ecosystem service maps zonal statistics for the Eddleston catchment

Eddleston catchment

SENCE look up table scores		Biodiversity			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	976.65	1542.63	12.14	19.17
50-150	Medium	1625.3	3006.08	20.20	37.36
150-250	High	2453.79	1983.21	30.50	24.65
Above 250	Very high	2990.52	1514.34	37.17	18.82
	Total	**8046.3**	**8046.3**	**100**	**100**

SENCE look up table scores		Pollination			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	501.35	875.11	6.23	10.88
50-150	Medium	2436.10	3802.49	30.28	47.26
150-250	High	3048.50	2030.78	37.89	25.24
Above 250	Very high	2060.33	1337.90	25.61	16.63
	Total	**8046.3**	**8046.3**	**100**	**100**

SENCE look up table scores		Vegetation carbon			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	2004.72	742.32	24.91	9.23
50-150	Medium	1703.44	2172.76	21.17	27.00
150-250	High	2603.44	2018.35	32.36	25.08
Above 250	Very high	1734.68	3112.85	21.56	38.69
	Total	**8046.3**	**8046.3**	**100**	**100**

SENCE look up table scores		Land erosion risk			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	10.17	12.02	12.64	14.94
50-150	Medium	15.93	36.08	19.80	44.84
150-250	High	33.05	21.76	41.07	27.04
Above 250	Very high	21.32	10.61	26.49	13.19
	Total*	**80.5**	**80.5**	**100**	**100**

*: Small total catchment area due to coarse scale (1:250 000) input soil data used in mapping this ecosystem service

SENCE look up table scores		Soil carbon storage			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	1376.93	1231.69	17.11	15.31
50-150	Medium	1509.17	2718.24	18.76	33.78
150-250	High	1794.29	2027.75	22.30	25.20
Above 250	Very high	3365.87	2068.58	41.83	25.71
	Total	**8046.3**	**8046.3**	**100**	**100**

SENCE look up table scores		Water quality regulation			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	1150.79	1106.62	14.30	13.75
50-150	Medium	1860.86	3245.36	23.13	40.33
150-250	High	3021.63	2296.96	37.55	28.55
Above 250	Very high	2013.00	1397.34	25.02	17.37
	Total	**8046.3**	**8046.3**	**100**	**100**

SENCE look up table scores		Flood control			
Relative values	Qualitative scale	Area (ha)		Percentage (%)	
		1946	2009	1946	2009
Below 50	Low	1281.54	187.86	15.93	2.33
50-150	Medium	749.94	520.81	9.32	6.47
150-250	High	3992.56	2615.73	49.62	32.51
Above 250	Very high	2022.25	4721.86	25.13	58.68
	Total	**8046.3**	**8046.3**	**100**	**100**

NB: Crops, livestock and timber production ecosystem servcies not included as zonal statistical counts were not done for these.